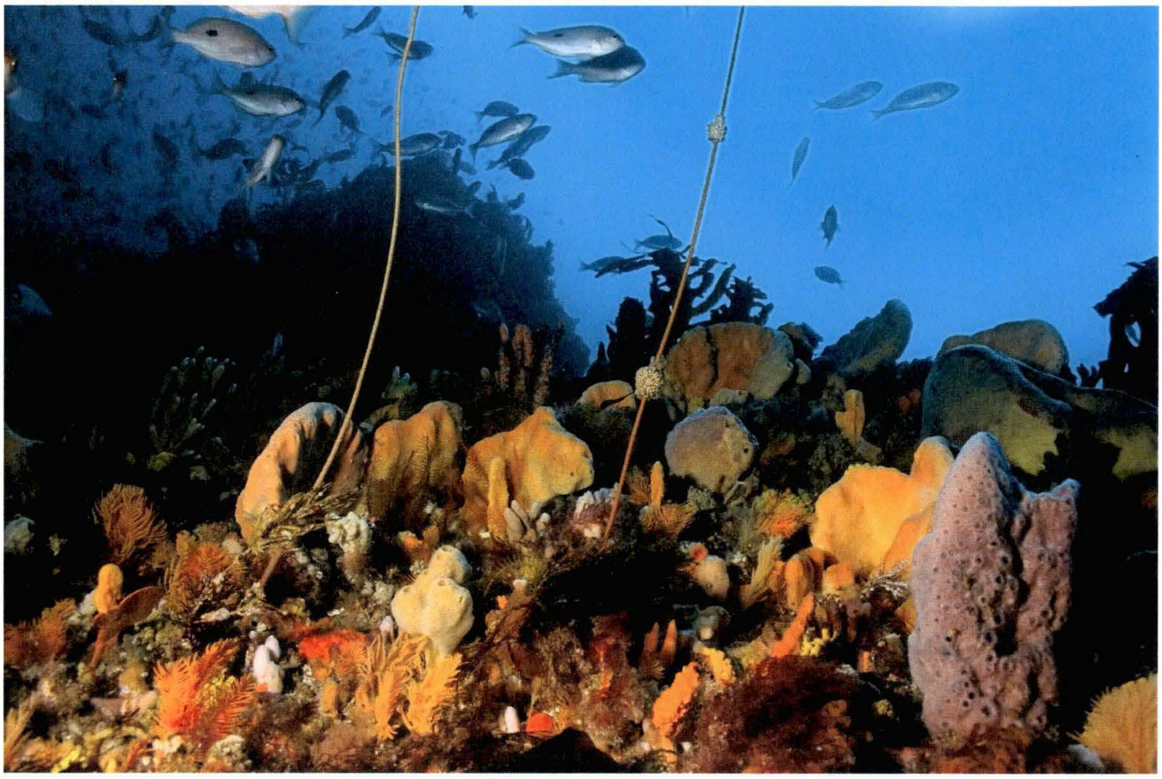
QUALITATIVE MODELLING TO AID ECOSYSTEM ANALYSES FOR FISHERIES MANAGEMENT IN A DATA-LIMITED SITUATION



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ABSTRACT

The need for ecosystem-level analyses is becoming widely recognised following the failure of single-species management in many systems around the world. In order to include multispecies and environmental interactions, ecosystem based fisheries management (EBFM) (known as Ecosystem Approach to Fisheries or EAF in Europe), has been put forward as a necessary next step for the management of fisheries. Policy regarding EBFM has been produced in Australia, the North Sea and the North Pacific yet the practical application of the approach has not been undertaken in any region. The implementation of EBFM has been slow due to the complexity of ecosystems and the large data needs usually required for ecosystem-level analyses. This study investigates a process of ecosystem analyses with the aim of aiding the swift implementation of EBFM.

The banded morwong (*Cheilodactylus spectabilis*) fishery was investigated in this case study due to the large number of non-target species captured in addition to the target species. The area in which this fishery operates is also important as it is subject to a number of other fisheries, increasing sea surface temperature (SST) and the influx of a number of warm-water invasive species.

Ecosystem analyses were undertaken in a number of steps. First, the definition of the spatial scale of the ecosystem and its constituents was undertaken using biodiversity survey and commercial catch sampling data. Multivariate statistics were used to identify spatial differences in fish, invertebrate and macroalgal communities between and within the area in which the majority of banded morwong fishing occurs. Significant differences in community composition suggested the spatial bounds of the

inshore reef ecosystem should occur at Eddystone Point (Lat. 40.99, Long. 148.35) and south Bruny Island (Lat. 43.27, Long 146.99). Commonly captured species were identified using commercial catch sampling data. The species with the highest catch were identified as banded morwong, blue throat wrasse, purple wrasse, draughtboard shark, marblefish, bastard trumpeter and long-snouted boarfish. Commonly sighted fish, invertebrate and algal species were identified using biodiversity surveys. Correlations with environmental data were investigated for invertebrate and algal species and identified a significant positive trend between sea surface temperature and the invasive urchin *Centrostephanus rodgersii*. A number of data gaps, such as the need for trophic information, and trends for further analysis were also highlighted.

The second step undertaken in these analyses was the production of qualitative models to examine the dynamics of the banded morwong fishery and the formation of urchin barrens on the inshore reef ecosystem. Profit was found to destabilise the fishery, while a total allowable catch (TAC) had a stabilising impact on the fishery, profit and stock biomass. Rock lobster were predicted to be able to control urchin abundance and limit barren formation. Qualitative predictions were generated from each model to guide and focus further research using quantitative models.

A survival experiment focussing on the target and non-target species captured in the banded morwong fishery was undertaken as the third step in the ecosystem analyses. This survival experiment was used to relate fish condition at capture to overall fishery mortality (including retained and discarded individuals) and discard survival rates. The majority of banded morwong (97.8%) were found to survive for the period of study (seven days). The species with the highest fishery-induced mortality and the highest discard mortality were the commercially valuable non-target species, blue throat wrasse

(86.21% fishing-induced mortality) and long-snouted boarfish (34.67% discard mortality). This result suggests the banded morwong fishery may have a negative impact on the commercial wrasse fishery, which also operates in the inshore reef ecosystem of eastern Tasmania. Fish condition at capture was suggested as a proxy for mortality in banded morwong and may be used to reduce waste and increase profits in the commercial fishery.

The fourth step in the process was the collection of ecosystem-specific dietary information for six commercially important fish species. Banded morwong, bastard trumpeter and wrasse (*Notolabrus* spp.) were found to feed primarily on benthic invertebrates (>53% IRI). In contrast, marblefish were found to be herbivorous (91% IRI) while long-snouted boarfish fed almost exclusively on ophiuroids (97.3% IRI). This information was used to create a detailed qualitative trophic model of the ecosystem and to examine the impact of model simplification and aggregation error on model results. Three methods of aggregation (Bray Curtis similarity, Euclidean distance and Regular Equivalence) were used to simplify the detailed trophic model. The aggregation of variables may create error as information may be lost between detailed and simplified models. The level of aggregation error produced by each method was calculated and assessed with regard to the simplification of future ecosystem models. The use of regular equivalence was found to produce the least amount of error (14%) between the simplified and detailed models.

The production of an Ecopath with Ecosim model and the comparison of results between quantitative and qualitative models was the final step in the process of ecosystem analysis. The model was produced using the trophic and survival information from the previous chapters. The impact of a TAC on the banded morwong fishery,

decreased macroalgal biomass and an increased biomass of urchins were investigated. The proposed TAC was found to be too large to allow banded morwong stocks to increase. Similar to the qualitative models (Chapter 3), rock lobster controlled urchin abundance and allowed foliose algal biomass to increase. The majority of results were found to be robust to the uncertainties and assumptions of the models. Inconsistencies were found to be due to differences in model building or the calculation of predictions.

There is a need for an efficient process of ecosystem analysis that increases the understanding of ecosystem dynamics and can be used in relatively data-poor situations. The method of analysis undertaken in this thesis, from ecosystem definition to qualitative modelling, data collection and quantitative modelling, can be of particular use in the implementation of EBFM in data-poor situations.

DECLARATION

Statement of originality

This thesis contains no material which has been accepted for a degree or diploma by the University or any other institution, except by way of background information and duly acknowledged in the thesis, and to the best of my knowledge and belief no material previously published or written by another person except where due acknowledgement is made in the text of the thesis, nor does the thesis contain any material that infringes copyright.

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14 September 2009

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STATEMENT OF CO-AUTHORSHIP

Chapters 2-4 and 6 in this thesis have been prepared as scientific manuscripts. In all cases the experimental design, research program, data analysis, interpretation of results and manuscript preparation were the primary responsibility of the candidate; however, chapters were undertaken in consultation with supervisors.

Chapters 2-4 and 6 Dr. J. Lyle and Dr. J. Dambacher were supervisors for this PhD project and, as such, contributed advice with regard to experimental design, data analysis, modelling and manuscript preparation.

Chapter 5 Sarah Metcalf (75%), Dr. Jeffrey Dambacher (15%), Dr. Alistair Hobday (5%) and Dr. Jeremy Lyle (5%). Dr. Jeffrey Dambacher was the Co-supervisor for this Ph.D. and contributed to the development of ideas and the manuscript preparation. Dr. Alistair Hobday and Dr. Jeremy Lyle (Primary Supervisor) both provided assistance with manuscript preparation.

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1. GENERAL INTRODUCTION

Many fisheries worldwide are in decline and classed as over- or fully-exploited (FAO 1997). The increasing human population and demand for marine resources determines that stress on fish stocks is likely to continue to increase. As all species are linked through trophic or non-trophic interactions (Vasas and Jordan 2006), the harvest of individual fish species needs to be examined in an ecosystem context. The need for ecosystem analyses in fisheries management is becoming widely recognised following the failure of single-species management in many systems around the world (Pikitch et al. 2004). As a result, ecosystem-based fisheries management (EBFM) or the European term, ecosystem approach to fisheries (EAF), is currently being developed for use in Australia and worldwide.

EBFM has been defined by Marasco et al. (2007) as 'management that recognises the physical, biological, economic and social interactions among the affected components of the ecosystem and attempts to manage fisheries to achieve multiple, often competing social objectives'. In this context, an ecosystem is a unit that includes all biotic parts of a community (i.e. food web, habitat) and the abiotic factors (i.e. climate, oceanography) that affect it (Tansley 1935, Link 2002). EBFM aims to protect ecosystem resilience to change in ecological, social or economic components as well as protect endangered species, essential habitat and bycatch (Pope et al. 2000, Pikitch et al. 2004). As yet, a formally accepted methodology for the application of EBFM has not been produced (Marasco et al. 2007). Nevertheless, calls for the swift implementation of EBFM have been made, as the potential benefits of implementation are greater than the risk of inaction (Pikitch et al. 2004).

In Australia, policy regarding EBFM is being developed by the Australian Fisheries Management Authority (www.afma.gov.au, Smith et al. 2007). Ecologically Sustainable Development (ESD), which is similar to EBFM, and policy frameworks for EBFM have also been developed in the north-east Atlantic and North Pacific (Garcia and Cochrane 2005). Yet, progress in the development and implementation of EBFM and ESD has been slowed by ecosystem complexity as a result of the number of species and potential perturbations that may be included in analyses. In addition, the large amount of data that is necessary to adequately represent all ecosystem components (e.g. species and perturbations) in quantitative models is often unavailable. In practice, many developing countries lack the means to undertake large-scale data collection and ecosystem analyses (Danielson et al. 2005, Wang et al. 2008). Yet, the lack of data and a means of data collection are not restricted to developing countries. Research priorities in developed countries are often rated by the monetary value of the resource and, as such, funds may be unavailable for investigating low-value fisheries and associated ecosystems. These data-poor fisheries may have a significant impact on ecosystems (Cheung and Sadovy 2004) and should not be overlooked by governments and researchers.

To allow the analysis of complex ecosystems, methods of ecosystem modelling that can be used to simplify systems for the prioritisation of data collection and detailed analyses are beneficial. Models that simplify ecosystem analyses can enable a greater understanding of whole ecosystem processes than may be possible without such modelling. Simplification can be useful in both data-poor and data-rich ecosystems with the former requiring simple models due to a lack of data and the latter having the potential to be too complex to tease out the factors driving ecosystem dynamics (Lawrie 2008). In addition, the use of simple models may reduce the difficulty of communication with stakeholders and decision-makers (Heemskerk et al. 2003). Qualitative modelling may be used to simplify systems to their core features (or species) and may be beneficial to guide and focus further ecosystem analyses (Levins 1974). This technique uses signed directed graphs (signed digraphs) to represent ecosystem structure and requires only the signs of interactions (positive, negative or no impact) between species or groups. As no quantitative data is necessary, qualitative modelling allows the investigation of ecosystem dynamics with limited data. The capacity to incorporate non-quantitative information, such as trends, into qualitative ecosystem models is also beneficial to allow the use of all available information in data-poor situations (Mace 1996). For instance, information on many ecological, social and economic groups may be restricted to general trends (i.e. increase or decrease), which can be incorporated into qualitative analyses that predict the direction of change in an ecosystem (increase, decrease or no change). Puccia and Levins (1985) suggested these qualitative predictions were valuable because a reliance on the quantification of interactions ignores the fact that the system is driven in large part by the structure and linkages within the system, not the absolute values of interactions.

Despite the potential advantages of qualitative modelling in EBFM, this technique has not reached the same level of acceptance as quantitative ecological models. This may be due to several inherent factors of qualitative analysis that are often seen as problematic by researchers, specifically the inability to make quantitative predictions from qualitative models may be seen as a significant disadvantage (Levins

1998). In general, scientific processes are based on quantitative analyses and many researchers find moving away from empirical models and studies difficult (Puccia and Levins 1985). In addition, the simplification of ecosystems to their core features may be difficult for researchers accustomed to highly detailed, complex models. Yet, quantitative models also have limitations and involve many assumptions. For instance, similar to qualitative models, the widely used quantitative ecosystem model, Ecopath with Ecosim (Walters et al. 1997), assumes the system is at or near equilibrium. Examples of other, often data-intensive, quantitative ecosystem models are Multispecies Virtual Population Analysis (MSVPA, Gislason and Helgason 1985), agent-based models (Carpenter et al. 1999) and biogeochemical ecosystem models (Fulton et al. 2004). All models, including these data-intensive quantitative models, assume the input data provide a reasonable representation of the real world and all necessary functional groups have been included. An important limitation in these models is the amount of data required for model production (Kelly and Codling 2006), which typically restricts their application in data-poor situations.

The use of qualitative models in conjunction with alternative modelling methods (i.e. quantitative models), may overcome many of the limitations of each approach while preserving the benefits, such as the ability to predict responses to change. Levins (1966) suggested models have three attributes that are important for investigating ecological systems: realism, precision and generality. A model that is precise produces point predictions for different output parameters (Orzack and Sober 1993), while the assumptions of a realistic model more closely resemble the actual process being studied (Nagy et al. 2007). Finally, a general model represents the dynamics of systems in the majority of cases (Nagy et al. 2007). All of these attributes cannot be maximised simultaneously and different types of models have therefore evolved to emphasise different attributes (Fig. 1.1).

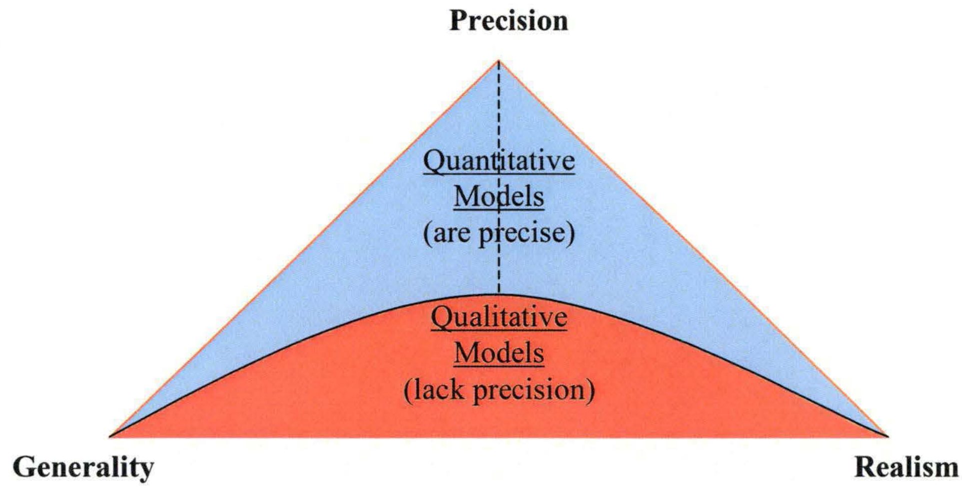


Figure 1.1 Conceptual diagram adapted from Levins' (1966) proposal on model building and its limitations. Realism, generality and precision cannot be maximised simultaneously and models must forgo one attribute to maximise the remaining two. Quantitative models are precise and either general or realistic (dashed line), while qualitative models are general and realistic but are not precise.

Quantitative models are used to provide precise predictions of the magnitude of change following a perturbation and may also be either general or realistic. Ecopath with Ecosim is commonly used in fisheries science and is an example of a quantitative model that sacrifices generality for realism and precision. In contrast, qualitative models sacrifice precision for realism and generality (Levins 1966). The magnitude of change due to perturbation cannot be obtained through qualitative modelling because only the signs of interactions (positive, negative, or no impact) are used during model construction. Qualitative models can be beneficial in order to step back from the complexity of data-intensive models and instead focus on the structure of the system, and the processes that drive system dynamics. Qualitative and quantitative models can be used in conjunction to provide information on different aspects within the ecosystem. For instance, the use of a general and realistic approach, such as qualitative models, may be beneficial to gain an initial understanding of the system and its dynamics. These models may be used to highlight issues and prioritise further research and once this understanding has been achieved, precise quantitative models could be used to increase knowledge and predictive capacity. The use of multiple models can also be beneficial as the predictions of one model alone may not be reliable due to model assumptions and uncertainties. Consistent results from models with diverse assumptions and uncertainties determine that the prediction may be robust to these differences (Levins 1993). A process, utilising multiple modelling types that allows the prioritisation of data collection and the guidance of further research may be useful in data-limited situations to provide a starting point for ecosystem analyses and quantitative modelling. As a result, both qualitative and quantitative models are explored in this thesis to investigate a process through which ecosystem analyses may be undertaken to assist the implementation of EBFM in a data-limited situation. This study is of particular importance in data-poor situations if such fisheries are to be managed sustainably and to identify methods that can assist the swift implementation of EBFM. To investigate the process through which ecosystem analyses may be undertaken, a relatively small live-fish fishery in Australia and its associated ecosystem were used as a case study.

The live-fish fishery for banded morwong (*Cheilodactylus spectabilis*) operates on the inshore rocky reefs of Tasmania and Victoria, Australia (Ziegler et al. 2008) with the majority of fishing occurring on the east coast of Tasmania (Fig. 1.2). This fishery utilises gillnets as method of capture and has recent annual catches in the order of 50t. The fishery may also have a wider impact on the supporting ecosystem through the capture of a range of non-target species (n = 45, J.M. Lyle pers. comm.). As a result, this fishery has the capacity to significantly impact the inshore reef ecosystem of eastern

Tasmania by changing the composition and abundance of a number of resident fish species. The banded morwong fishery is by no means the largest or most important commercial fishery on the east coast of Tasmania; however, since the instigation of the fishery in early 1990s, it has already had a significant impact on target populations. Banded morwong are long-lived (>90 years, Ewing et al. 2007, Fig, 1.3) and have experienced a decrease in age at maturity (from four years to three years) as well as accelerated growth rates (13% increase) since the commercial fishery began (Ziegler et al. 2007). Zeigler et al. (2007) hypothesised that these changes were the result of density-dependent compensation. Upper and lower size limits have been incorporated into management and are designed to provide protection to spawners. Nonetheless, it is clear that fishing has significantly altered the demographics of banded morwong populations in Tasmania, with stocks now dominated by young fish and a small number of age classes (Zeigler et al. 2007). Such demographic changes may be sustainable in the short-term, yet they suggest that the long-term sustainability of the fishery is questionable.

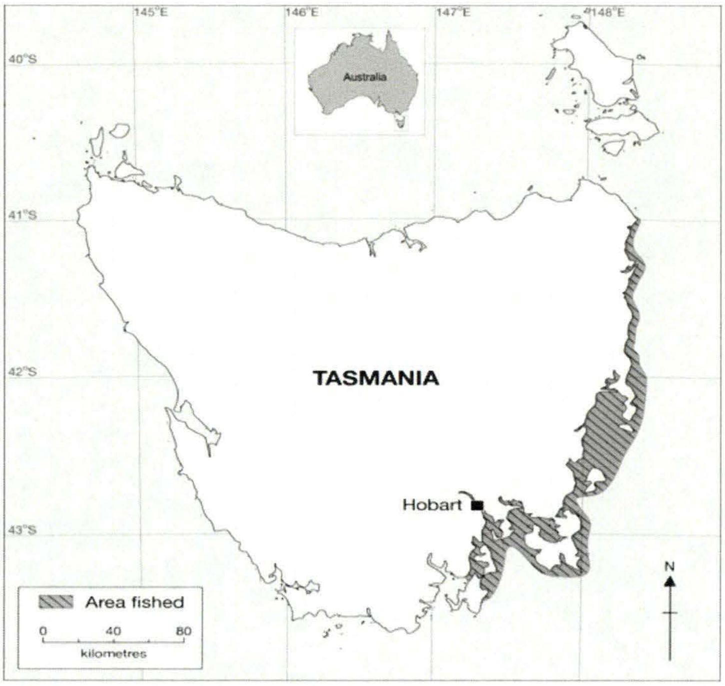


Figure 1.2 The east coast of Tasmania, Australia, where the majority of banded morwong fishing effort occurs (shaded).

Figure 1.3 Banded morwong (*Cheilodactylus spectabilis*) captured in the commercial live fish fishery.

A number of other commercial fisheries operate on the inshore reef ecosystem of eastern Tasmania and use a variety of techniques (i.e. pots, line, fish traps, dive collection). The fisheries for southern rock lobster (*Jasus edwarsii*) (368t from the east coast in 2006/07, Haddon and Gardner 2008) and blacklip abalone (*Haliotis rubra*) (488t in 2006/07, Tarbath et al. 2007) are significant due to their size (tonnes captured) and economic value. A number of smaller commercial scalefish fisheries also occur, including the live wrasse trap and line fishery (*Notolabrus* spp., approx. 55t in 2006/07, Zeigler et al. 2008) and a gillnet fishery for various scalefish species including bastard trumpeter (*Latridopsis forsteri*, approx. 20t in 2006/07, Zeigler et al. 2008). Recreational fishing is also popular on the east coast of Tasmania due to high population density, sheltered waters and the presence of many coastal access points. Southern rock lobster (Haddon and Gardner 2008), abalone (*Haliotis* spp.)(Tarbath et al. 2007), bastard trumpeter (*Latridopsis forsteri*) and blue warehou (*Seriolella brama*)(Lyle 2000) are examples of recreationally important species.

In addition to fishing, the inshore reefs of eastern Tasmania are subject to the impacts of invasive species, such as Japanese kelp (*Undaria pinnatifida*) and the long-spined urchin (*Centrostephanus rodgersii*). These species may cause significant disruption to ecosystem dynamics by utilising and competing for resources, such as space (Valentine and Johnson 2005) and the creation of urchin barrens (Hill et al. 2003). Natural environmental variability, as well as climate change effects, may also pose a threat to the sustainability and functionality of the inshore reef ecosystem and associated fisheries. Natural environmental variability has been blamed for variations in the recruitment of bastard trumpeter, a valuable non-target species (Murphy and Lyle 1999). The impacts of climate change on the recruitment success of this and other species could have potentially wide-ranging effects on the ecosystem. Increased sea surface temperature (SST) over the last decade on the east coast of Tasmania has been linked to climate change (Lyne et al. 2005). This increase in SST has been associated with an influx of warm water species (Hobday et al. 2006) and a range contraction of species restricted to cooler waters (Edgar et al. 2005). Tasmania is particularly vulnerable to changes in SST as some species in Tasmania's south rely on a seasonal influx of cold Sub-Antarctic mode water for nutrient-enhancement and increased phytoplankton growth (Prince and Griffin 2001). In recent years this current has been restricted to increasingly southern latitudes while the southerly reach of the warm nutrient-poor East Australian Current (EAC) has extended (Ridgway 2007). As fishing, invasive species, natural environmental variation and climate change may affect the inshore reef ecosystem in different ways, the assessment of multiple impacts on the ecosystem and individual species should be considered.

1.1 Thesis objectives

This thesis aims to investigate methods of analysis that could be used to assist the implementation of EBFM in data-limited situations. In order to achieve this aim, a key secondary aim was to identify ecosystem scale and constituents as well as highlight the ecosystem dynamics and perturbations that impact banded morwong populations, the live-fish fishery and the associated reef ecosystem. Therefore, the rationale for the structure of the thesis into five data/modelling chapters (Chapters 2-6) was to allow the identification of critical factors, such as ecosystem scale and fishery-induced mortalities. The banded morwong fishery and its supporting ecosystem were selected as a case study due to the potentially large impact of the fishery on the ecosystem and target populations, in addition to a number of threatening processes (i.e. invasive species and climate change) that may impact the region. The investigation was undertaken in a number of steps. First, the ecosystem was defined to ensure the appropriate species, spatial and temporal scale, were being investigated (Chapter 2). A number of data gaps, such as trophic information for a number of commercially important species, were also highlighted in this chapter. Secondly, predictions were produced from a number of qualitative models that focussed on the dynamics of the banded morwong fishery system and the impact of urchin grazing and rock lobster fishing (Chapter 3). Condition at capture and survival of a number of commercially important target and non-target species captured in the banded morwong fishery were then investigated (Chapter 4). This was undertaken to determine the overall impact of the fishery and to calculate discard survival rates. The next step involved the collection of trophic information for key species captured in the banded morwong fishery (Chapter 5). This dietary information was used to produce a qualitative model to investigate trophic linkages and the utility of different simplification and aggregation techniques. Finally, a quantitative Ecopath with Ecosim model was produced to investigate predictions of response to perturbations, such as increased fishing and reductions in primary production (Chapter 6). The role of qualitative models in guiding quantitative model building and the use of multiple models to examine the robustness of conclusions were investigated in this chapter. These steps were synthesised and discussed in relation to the benefits of this process in EBFM (Chapter 7). This research has implications for the management of ecosystems and fisheries elsewhere by assisting the implementation of EBFM.

2. DEFINING THE INSHORE REEF ECOSYSTEM AND CONSTITUENTS

2.1 INTRODUCTION

The analysis of the ecosystem impacts of fishing is becoming an important element of fisheries management (e.g. Trites et al. 1999, Bundy and Pauly 2001, Gasalla and Rossi-Wongtschowski 2004, Coll et al. 2007). This trend may be attributed to the recognition that changes in species composition and abundance, due to perturbations that are not directly related to the fishery (e.g. environmental change, fluctuations in primary productivity), may affect commercially important target species. Similarly, the impacts of fishing might not be restricted to target species and may have additional effects throughout the ecosystem through trophic and non-trophic links. The analysis of ecosystem impacts is usually undertaken using models that require the spatial scale and constituents to be defined prior to use. Failure to properly define the ecosystem scale and constituents can result in the creation of haphazard ecosystem bounds and may result in important relationships and species being excluded or misrepresented (Turner et al. 1989). The results and conclusions of ecosystem models may then be of little use for fisheries management.

The assessment of ecosystem spatial scale and the species impacted by the banded morwong (*Cheilodactylus spectabilis*) fishery was undertaken in order to construct ecosystem models with which to investigate fishery effects. This gillnet fishery operates largely on the inshore reefs off the east coast of Tasmania (Fig. 2.1) and is one of the major scalefish fisheries in Tasmania. Commercial fishing for banded morwong also occurs off the Bass Strait Islands and the Victorian coast; however, this study has focussed on the east coast of Tasmania where the majority of the effort and catch occurs (Ziegler et al. 2008). The inshore reefs, on which the fishery operates, fringe a large proportion of the coastline of eastern Tasmania and are dominated by temperate reef biota including a variety of macroalgae, such as giant kelp (*Macrocyctis pyrifera*) (Seamap Tasmania, www.utas.edu.au/tafi/seamap/). The wave exposure and tidal gradient in this region are variable with tides of around 2.5m in the north and 0.6m in the south (Edgar et al. 1999). In this region, banded morwong are subject to an annual catch in the order of 50 tonnes (Ziegler et al. 2008). In comparison to the catch of the Tasmanian rock lobster (*Jasus edwardsii*, 2006/7 catch: 368t, Haddon and Gardner 2008) and blacklip abalone (*Haliotis rubra*, 2006/7 catch: 488t, Tarbath et al. 2007) from the east coast of Tasmania, production from the banded morwong fishery is relatively small. In addition to commercial fisheries, the inshore reefs are also heavily exploited by recreational fishers. Recreational fishers generally use gillnets, which are permitted under license in Tasmania, and in 2007 there were in excess of 9550 recreational net licences issued (Ziegler et al 2008). Many of these recreational fishers also target fish associated with the inshore reefs of eastern Tasmania, such as bastard trumpeter (*Latridopsis forsteri*) and blue warehou (*Seriolella brama*). While the effects of the banded morwong fishery on the target species have been significant, with increased growth rates and decreased age at maturity suggested to be due to fishery-induced effects on population density (Ziegler et al. 2007), there are also many other factors that may influence the inshore reefs.

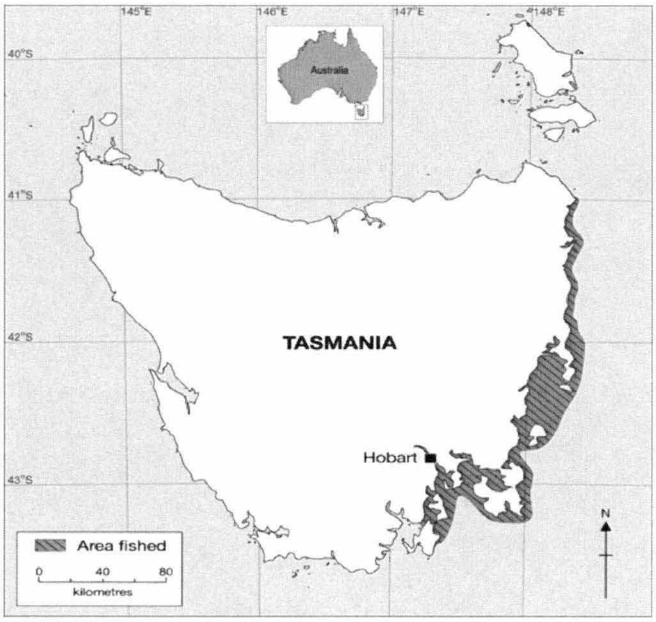


Figure 2.1 Area (shaded) in which the majority of banded morwong (*Cheilodactylus spectabilis*) fishing effort occurs in Tasmania.

The banded morwong fishery may have broader impacts on ecosystem dynamics through the capture of a diverse range of non-target species (J.M. Lyle, unpublished data). The composition and spatial variability in the catch rates of non-target species captured in the banded morwong fishery have not been previously investigated and may provide an indication of overall fishing effects on the inshore reef ecosystem. Many of the reef-associated species in Tasmania, including banded morwong and wrasse (*Notolabrus* spp.), have high site fidelity and fishing for these species may therefore cause localised depletion (Barrett 1995, Murphy and Lyle 1999). Such depletions have unknown consequences within the reef ecosystem. In addition, fisheries with a high rate of non-target species capture may negatively impact the catch and profits of other fisheries. This can occur if byproduct species, which are the target of other fisheries, are retained or die following discarding. In this case, byproduct species are non-target fish of value that are retained by fishers. Whether the banded morwong fishery does have a negative impact on the catch (and profit) of other fisheries through the capture of byproduct species is unknown. To gauge such effects, knowledge of the composition and spatial variability of non-target catches is essential.

The inshore reef ecosystem off eastern Tasmania is also impacted by invasive species and environmental change. The spatial bounds and constituents of the ecosystem need to reflect these impacts in order to provide useful results for the management of the banded morwong fishery and the ecosystem. Tasmanian marine ecosystems have been perturbed by the most rapidly changing sea surface temperature (SST) in Australia, with a rise of 1 °C occurring over the last 60 years (Lyne et al. 2005). This increase is thought to be responsible for the influx of warm water species not previously found in the area. Originally from New South Wales, the long-spined urchin, *Centrostephanus rodgersii*, has been moving progressively southward, creating urchin barrens; areas devoid of foliose algae as a result of grazing by urchins. Urchin barrens are currently present on the north-eastern islands and east coasts of Tasmania (Johnson et al. 2005) and while some crustose algae may remain in the barrens, the loss of foliose macroalgae has been shown to decrease the abundance and diversity of species in the area (Willis and Anderson 2003, Graham 2004). The native urchin, *Heliocidaris erythrogramma*, is also known to create barrens on the east coast of Tasmania (Valentine and Johnson 2005). Increasing *C. rodgersii* abundance, in addition to grazing by *H. erythrogramma*, may have a significant impact on the Tasmanian reef ecosystem due to an increased occurrence of urchin barrens (Johnson et al. 2005, Ling 2008). Another species that plays a significant role in the ecosystem is the giant kelp, *M. pyrifera*. This kelp provides an important habitat for many fish and invertebrates

(Anderson 2001). In recent years the distribution and abundance of *M. pyrifera* has been variable with extensive die back in some areas (Sanderson 1997) that has been attributed to natural variability in addition to increasing SST (Eddyvane 2003). Some kelp bed regeneration has been observed in marine reserves off the south and south-west coasts (N. Barrett, pers. comm.). This regeneration is thought to be due to the recovery of herbivore predators within the reserve, which reduces the consumption kelp and allows regeneration (Edgar and Barrett 1999, Shears and Babcock 2003).

The purpose of the present study was to spatially define the ecosystem and identify key constituents that may be affected by the banded morwong live fish fishery and various threatening processes, such as an influx of invasive species. In particular, commercially important species, common and invasive species (constituents) including invertebrates and macroalgae, were highlighted. In addition, the investigation of interspecies and environmental correlations was undertaken.

2.2 METHODS

2.2.1 Data types and study locations

Data sets

Two data sets were available for the investigation of the spatial bounds and constituents of the focal ecosystem. These data were obtained during biodiversity surveys and commercial catch sampling. Additional analyses were undertaken using environmental (SST) data.

Biodiversity survey data

Transect based dive surveys were conducted at a number of sites around the Tasmanian coast to investigate marine biodiversity and community structure. This data, provided by G. Edgar and N. Barrett (Marine Research Laboratories, Tasmanian Aquaculture and Fisheries Institute), consisted of surveys undertaken at 17 locations around the Tasmanian coast and comprised a total of 118 sites (Fig. 2.2, Table 2.1). All data, including data from locations on the north, west and south coasts, were used in the analyses to ensure the spatial bounds were appropriate for further ecosystem investigations.

The biodiversity survey data included species composition and abundance for fish, invertebrates and macroalgae (percent cover) as well as rugosity at each site. Data were collected along transect lines, though the area surveyed differed for fish, invertebrates and algae as follows (Edgar and Barrett 1997):

- Mobile fish- abundance was based on four 50m×5m transects while swimming at a constant speed above the algal canopy;
- Invertebrates and cryptic fish- abundance was based on four 50m×1m transects searching above and below the algal canopy;
- Algae- percent cover was based on ten 0.5m^2 quadrats abutting the 50m×1m transect line at 10m intervals with all algal species within the quadrat identified.

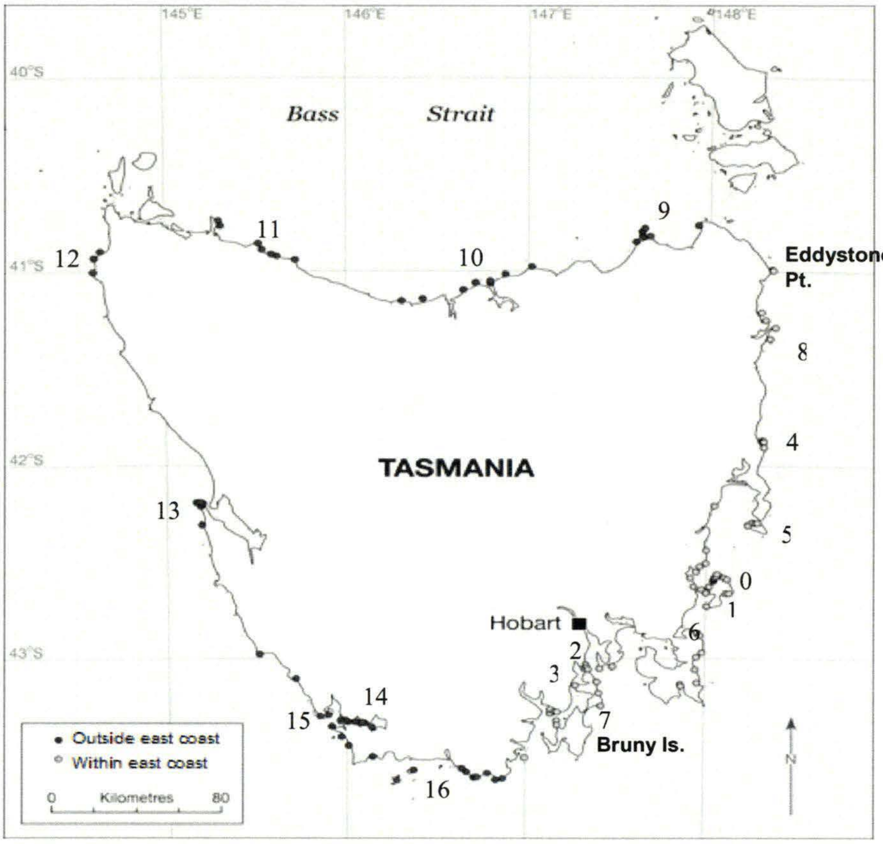


Figure 2.2 The location of biodiversity surveys. Numbers refer to broad locations (n = 17) (see Table 2.1) while each circle represents an individual survey site (n=118) within the location. See section 2.2.2. for a description of 'Outside the east coast' and 'Within the east coast'.

Abundance data was converted to number per unit area for the analyses. Dive transects were limited to the 5m depth contour with individual locations sampled over one to 11 years during spring and autumn from 1992 to 2002 (Table 2.1).

Table 2.1 Biodiversity survey locations and details for data collected between 1992 and 2002. See section 2.2.2. for a description of the 'east coast' region.

Location number	**Within the 'east coast' region**	**Location name**	**Total no. years surveyed**	**Years surveyed**	**No. sites**
0	Yes	Maria Island (north)	11	1992-2002	9
1	Yes	Maria Island (south-west)	11	1992-2002	7
2	Yes	Tinderbox	10	1992-1997, 1999-2002	5
3	Yes	Ninepin Point	10	1992-1997, 1999-2002	5
4	Yes	Bicheno	8	1992-1994, 1997, 1999-2002	3
5	Yes	Shouten Island	5	1992-1994, 1997, 2000	4
6	Yes	Tasman Peninsula	1	1994	8
7	Yes	Bruny Island	1	1994	4
8	Yes	St. Helens/ Eddystone Point	2	1994, 1999	7
9	No	North Coast	3	1992,1995,1999	6
10	No	North	3	1994-1995, 1999	8
11	No	North-west	5	1992-1995, 1999	7
12	No	North-west coast	2	1994-1995	3
13	No	Central-west	1	1994	6
14	No	Bathurst Harbour	1	1993	6
15	No	Port Davey	4	1993-1994,1997-1998	17
16	No	South-west	1	1994	13

Commercial catch sampling data

The second type of data used for the analysis of ecosystem spatial scale and constituents was commercial catch sampling data. This sampling was undertaken to determine the catch and size composition of target and non-target species in the banded morwong commercial fishery. The sampling was supplemented by research fishing during the spawning season and designed to collect biological samples for the assessment of population characteristics (e.g. Murphy and Lyle 1999, Ziegler et al. 2007). All sampling occurred onboard commercial vessels using standard commercial fishing practices such as set times, depth of set and gillnet mesh sizes (115 to 140 mm). The number and length of nets set per day was selected by the operator and was variable according the prevailing sea conditions and extent of gear interference by seals (Australian fur seal, *Arctocephalus pusillus*, and New Zealand fur seal, *Arctocephalus forsteri*).

Sampling events were generally opportunistic and did not have equal replication between years and locations. Commercial catch sampling records were available from 1994-1997 and 2001-2003, and contained information such as date, location, number of net sets, catch composition by species and numbers caught (including target and non-target species). No sampling occurred during 1998-1999 and sampling was limited to a single day in 2000 while a maximum of 32 fishing days were sampled over 10 months in 1995.

The spatial extent of sampling was restricted to the east coast of Tasmania between South Bruny Island (43.5 S 147.1 E) in the south-east and Cape Barren Island (40.5 S 148.4 E) in the north-east, reflecting the main focus of fishing effort. For ease of analysis, individual sampling locations were allocated to one of four regions on the east coast of Tasmania since specific Global Positioning System (GPS) locations were not always available (Fig. 2.3). Regions 5-7 were sampled in each of the years for which data were available with the exception of 2000, when sampling was limited to region 6. Sampling also occurred in region 4 in 1996 and 2002. Investigations into spatial differences in catch composition and species abundance were undertaken using available data from all sampled years, excluding 2000.

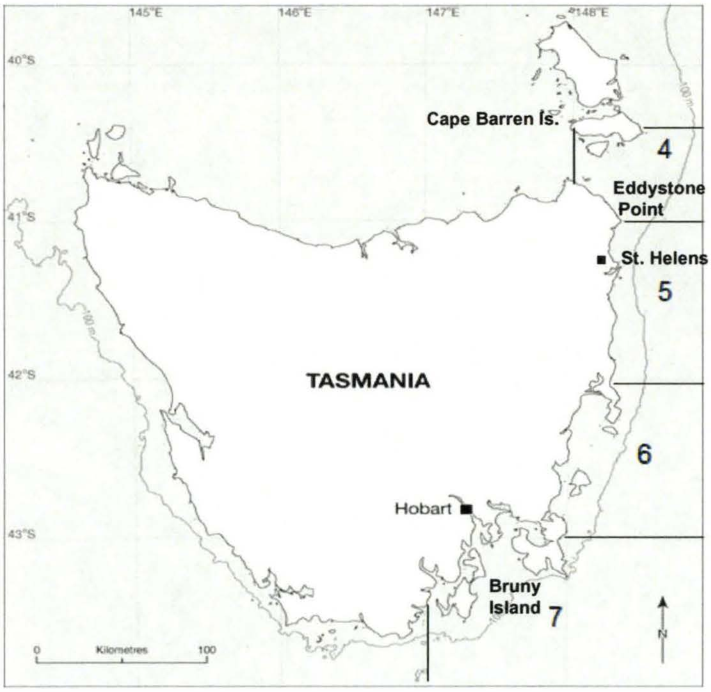


Figure 2.3 Sampling regions (4-7) for commercial catch sampling records.

2.2.2 Data analysis

Defining the ecosystem: Spatial differences

Community composition around Tasmania

The first step in the definition of the ecosystem was an examination of differences in the spatial heterogeneity of reef communities on the east coast, where the majority of banded morwong fishing occurs, and communities elsewhere around Tasmania (Table 2.1). For these analyses, the east coast was defined as the region between Eddystone Point (Lat. 40.99, Long. 148.35) and south Bruny Island (Lat. 43.27, Long 146.99) (Fig. 2.3). Differences in fish, invertebrate and macroalgal communities on the east coast and elsewhere around Tasmania were assessed by examining species abundance and composition using the biodiversity survey data. Analysis of Similarities

(ANOSIM) based on Bray-Curtis dissimilarities were used as this metric is not influenced by joint-absences (i.e. multiple species abundances of zero), which were common in the data sets. All the data were square-root transformed prior to analysis using Primer 5 (Version 5.2.9) to dampen the effects of dominant species. A lack of difference between communities would indicate that the communities on the east coast were the same as elsewhere in the state. This result would mean that the east coast, where the majority of banded morwong fishing occurs, could not be considered a separate and discrete ecosystem. As a result, ecosystem analyses would need to be undertaken on a broader spatial scale.

Community composition within the east coast of Tasmania

Both biodiversity and commercial catch sampling data were used to investigate spatial differences in fish composition within the east coast region. This was undertaken to examine whether there were differences in overall species observations or catch composition within the east coast that would be indicative of a region too dissimilar to be considered a single ecosystem. Fish were the sole group used in these analyses as the banded morwong fishery was the primary focus of the study. The commercial catch sampling data were used to examine whether catch composition in this region was spatially heterogeneous while the biodiversity data was used to test for spatial differences in the overall reef fish community structure on the east coast. These analyses were undertaken using ANOSIM.

Defining the ecosystem: Species and environmental trends

The second step in the definition of the ecosystem was to determine the most frequently sighted and captured species within this area. Both the commercial catch sampling and biodiversity survey data were used to undertake these analyses. The identification of the most frequently sighted and captured species was undertaken because these species may be common and play a large role in ecosystem function through trophic and non-trophic relationships (Schwartz et al. 2000).

Examination of interspecies relationships through correlations in abundance were undertaken to highlight trends that should be investigated using modelling. Biodiversity survey data from the east coast of Tasmania were used to perform these analyses. The abundance of two invasive species, the long-spined sea urchin (*C. rodgersii*) and the Japanese kelp (*U. pinnatifida*) as well as the native barren-forming urchin (*H. erythrogramma*) were investigated separately with regard to the giant kelp (*Macrocystis pyrifera*) distribution and abundance. This was important, as giant kelp is a potential prey of urchins and a competitor with Japanese kelp.

Correlations between SST and species abundance were also investigated using the biodiversity survey data to highlight trends for investigation using ecosystem models. SSTs were obtained for the east coast of Tasmania from St Helens (Lat. 41.33, Long. 148.27) to Bruny Island (Lat. 43.28, Long.147.28) (Fig. 2.3). SST data were downloaded using Spatial Dynamics Ocean Data Explorer (SDODE) from the CSIRO Marine and Atmospheric Research Laboratories (www.cmar.csiro.com.au). Weekly measurements were obtained from January 1994 to December 2003 to coincide with the years for which commercial catch sampling and biodiversity survey data were available. Data were restricted to within 100m from the coast in order to focus on the inshore reefs. SSTs were averaged (mean) within fishing regions (Fig. 2.3) to reduce the noise generally associated with environmental data (Smith and Reynolds 2003, Knutson et al. 1997). This stratification by SST was also undertaken because temperatures in the north and south of Tasmania vary substantially (www.bom.gov.au/sat/SST/sst.shtml). Stratifying SST by fishing region enabled the investigation of correlations between species abundance and SST.

Correlation with environmental factors was investigated for macroalgal and invertebrate abundances. Fish were excluded as their abundances can be substantially affected by the environmental conditions that occurred during relatively long larval stages (Vonherbing and Hunte 1991). In contrast, invertebrate and macroalgal abundances generally have shorter larval stages and dispersal times (Kinlan and Gaines 2003) that may more closely relate to short-term environmental conditions. In addition, between spawning and recruitment, fish may be subjected to numerous environmental (Williams and Quinn 2000) and trophic interactions (Purcell 1985, Meekan and Fortier 1996) that may confound these analyses. Correlations between mean invertebrate and algal abundances with SST between 1992 and 2002 were investigated. A number of species were investigated separately, including *C. rodgersii*, in addition to investigation into plant phyla and total invertebrate abundance. Pearson correlation was used to determine the level of correlation between species abundance and SST by fishing block (stratified by SST) on the east coast of Tasmania.

2.3 RESULTS

2.3.1 Defining the ecosystem: Spatial differences

Community composition around Tasmania

A significant difference in total species composition and abundance was observed between the east coast of Tasmania and elsewhere ($R = 0.951$, $p = 0.001$). In addition, both fish ($R = 0.968$, $p = 0.001$) and invertebrate ($R = 0.430$, $p = 0.001$) communities were significantly different between the east coast and the rest of Tasmania. In contrast, a high level of similarity was observed between algal communities ($R = -0.053$, $p = 0.796$). As a result of the significant differences in total species composition and abundance as well as fish and invertebrate communities, the reef communities of the east coast could be considered distinct from those elsewhere around Tasmania.

Community composition within the east coast of Tasmania

No distinct separation was evident in the catch composition among regions (Fig. 2.3) of the east coast ($R = 0.109$, $p = 0.078$) using the commercial catch sampling data. Similarly, no differences in fish species abundance and composition were uncovered ($R = -0.080$, $p = 0.605$) between east coast locations (Table 2.1) based on the biodiversity survey data. The area from Eddystone Point (Lat. 40.991, Long. 148.345) to south Bruny Island (Lat. 43.27, Long 146.996) was therefore deemed spatially homogenous in terms of fish community structure and defined as the inshore reef ecosystem of eastern Tasmania for this study.

2.3.2 Defining the ecosystem: Species and environmental trends

Common fish species and abundance

Biodiversity survey data

Several schooling species, namely barber perch (*Caesioperca razor*), mado (*Atypicythys strigatus*), blotch-tailed trachinops (*Trachinops caudimaculatus*) and common bullseye (*Pempheris multiradiata*) were found to have the highest abundances during the biodiversity surveys (Fig. 2.4). In addition, the commercially important purple and blue throat wrasses (*Notolabrus fucicola* and *N. tetricus* respectively) were frequently observed. Figure 2.4 is simply a summary of the most frequently observed species. Altogether there were 97 species sighted on the east coast of Tasmania (App. 1).

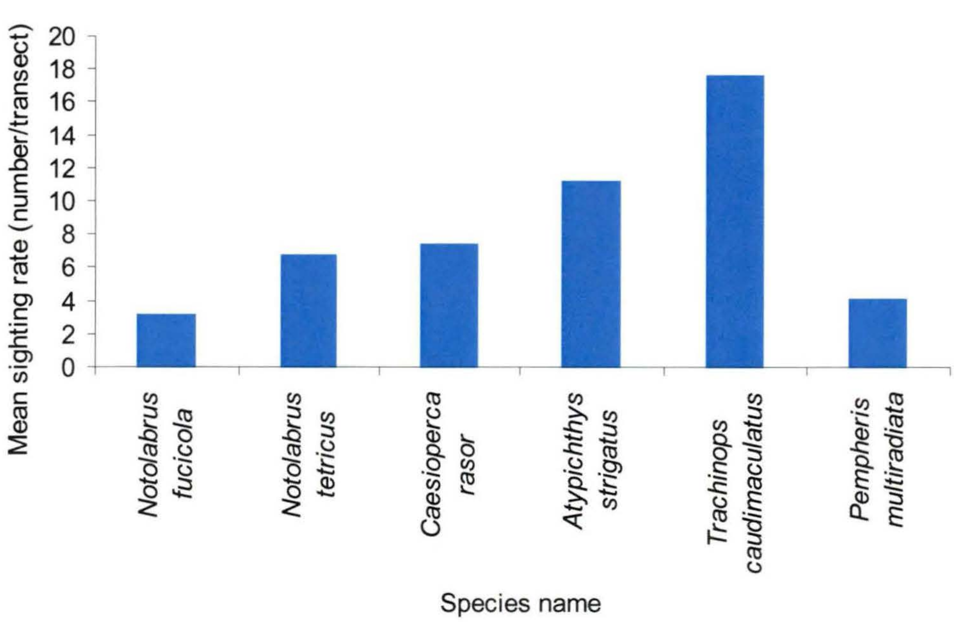


Figure 2.4 Mean sighting rates (number/transect) of frequently observed fish species on the inshore reefs of eastern Tasmania using biodiversity surveys between 1992 and 2002.

Commercial catch sampling data

Banded morwong was the most frequently caught species in the commercial catch sampling records and constituted 35% of the total catch (by numbers) in the monitored fishing operations (Fig. 2.5). The most common non-target species were: marblefish (*Aplodactylus arctidens*); draughtboard shark (*Cephaloscyllium laticeps*); long-snouted boarfish (*Pentaceropsis recurvirostris*); blue throat wrasse; bastard trumpeter (*Latridopsis forsteri*); and purple wrasse. These six non-target species collectively dominated catches, constituting 52.3% of the total catch by number. A further 39 species accounted for the remaining 13% of the catch. Four of the six major non-target species, namely blue throat wrasse, purple wrasse, long-snouted boarfish and bastard trumpeter, are commercially valuable (byproducts).

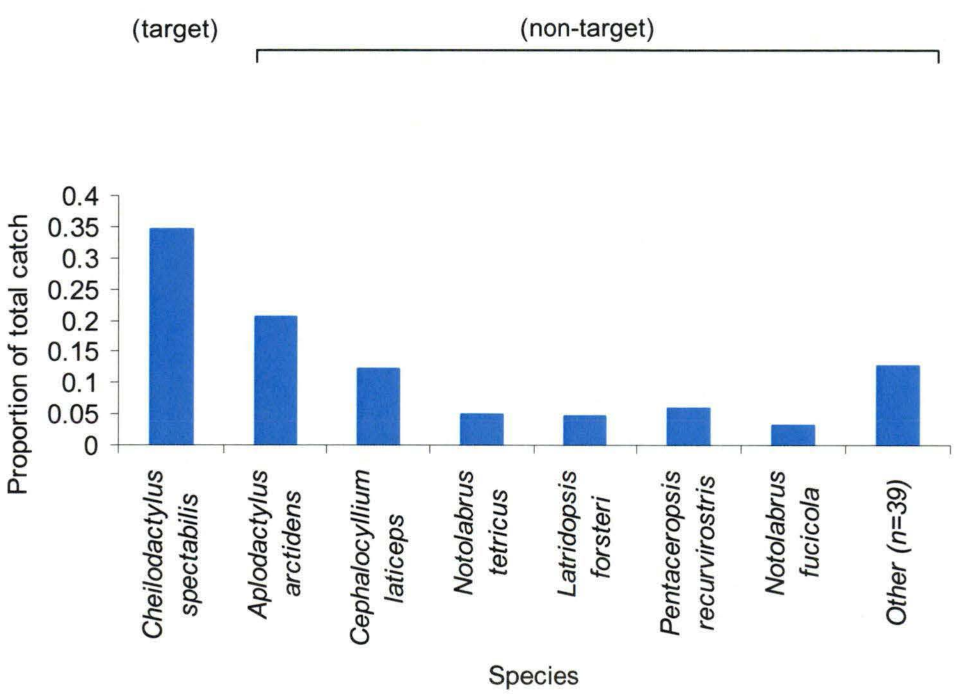


Figure 2.5 Proportion of the total catch (by number) of the banded morwong fishery between 1994 and 2004 based on commercial catch sampling of the major target and non-target species, as well as other species (n= 39).

Common invertebrate species and abundance

Biodiversity survey data

Altogether there were 73 species sighted on the east coast of Tasmania (App. 1); however, invertebrate abundance was heavily dominated by the crinoid, *Cenolia trichoptera*, and the echinoid, *Heliocidaris erythrogramma* (Table 2.2).

Table 2.2 The most frequently observed invertebrate species and their sighting rates from the inshore reefs of eastern Tasmania. Biodiversity survey data collected between 1992 and 2002 were used in these analyses.

Species	**Common name and/or Class**	**Sighting rate (number/transect)**
Cenolia trichoptera	Crinoidea	18.80
Heliocidaris erythrogramma	Echinoidea	16.46
Haliotis rubra	Blacklip abalone, Gastropoda	6.54
Cenolia tasmaniae	Crinoidea	4.52
Turbo undulatus	Periwinkle, Gastropoda	4.38
Goniocidaris tubaria	Echinoidea	2.90

Environmental trends

SST was not significantly correlated with the total invertebrate abundance (Pearson correlation coefficient = 0.535, $p = 0.112$) or the abundance of *C. trichoptera* (Pearson correlation coefficient = 0.225, $p = 0.533$) and *H. erythrogramma* (Pearson correlation coefficient = -0.197, $p = 0.585$) in any block. In contrast, investigation into the abundance of the invasive urchin, *C. rodgersii,* over time uncovered a gradual increase between 1992 and 2001 (Fig. 2.6). A positive correlation between *C. rodgersii* abundance and mean annual SST was found in region 4 (north east) (Pearson correlation coefficient = 0.786, $p = 0.002$) but not in region 5, possibly due to very low abundance in this region. In addition, there was a non-significant correlation between the potential competitors *H. erythrogramma* and *C. rodgersii* (Pearson correlation coefficient = -0.226, $p = 0.534$).

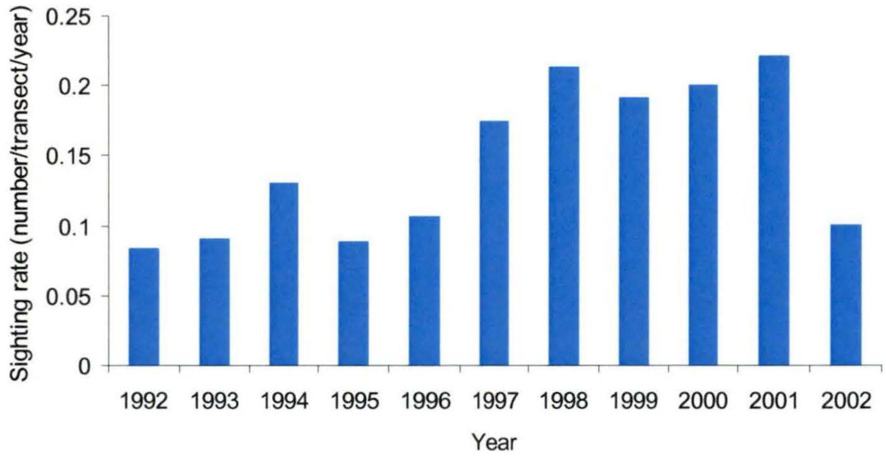


Figure 2.6 Sighting rate of invasive urchin *Centrostephanus rodgersii* on the east coast of Tasmania from biodiversity survey data.

Common macroalgal species and abundance

Biodiversity survey data

All locations displayed a high abundance of brown algae (P. *Phaeophyta*), in particular, *Phyllospora comosa* and *Ecklonia radiata* (Fig. 2.7). These two species were the most dominant algal species in the ecosystem; their abundance was an order of magnitude higher than the third-ranked species, *Carpoglossum confluens* (P. Phaeophyta). Other frequently sighted brown algal species were: *Cystophora retroflexa*; *Lessonia corrugata*; and *Seirococcus axillaris* (Fig. 2.7). Red algae (P. Rhodophyta), such as *Phacelocarpus pepperocarpus*, and green algae (P. Chlorophyta), such as *Caulerpa* spp., were also observed on the inshore reefs of eastern Tasmania; however, they comprised a smaller percentage cover than brown algae in the transects. The total number of algal species sighted was 87 (App. 1).

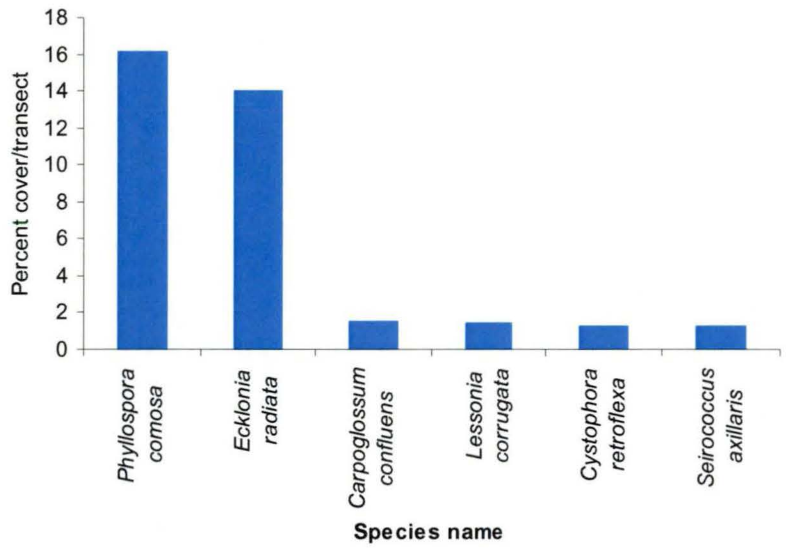


Figure 2.7 Average (mean) percent cover of the most abundant algal species from the biodiversity survey data on the inshore reefs of the east coast of Tasmania between 1992 and 2002. All species are brown algae (P. Phaeophyta).

Environmental trends

No significant correlations ($p > 0.05$) were observed between any algal phyla and SST for any block on the east coast of Tasmania. The percent cover of the invasive alga, *U. pinnatifida*, was highly variable during the period of study and a non-significant correlation between *U. pinnatifida* and *M. pyrifera* was found (Pearson correlation coefficient = 0.027, $p = 0.942$). Furthermore, the abundance of the urchin *H. erythrogramma* or *C. rodgersii* did not correlate with either algal species.

Table 2.3 Summary table showing the significance ($p < 0.05$) of all ANOSIM and correlations undertaken for spatial, environmental (SST) and inter-species relationships for the inshore reef ecosystem of eastern Tasmania.

Test	Significant?	Test	Significant?
East Coast vs. elsewhere around Tasmania-		Total invertebrates	N
All species	Y	Phaeophyta	N
Fish	Y	Chlorophyta	N
Invertebrates	Y	Rhodophyta	N
Macroalgae	N	*U. pinnatifida*	N
Spatial differences within the east coast ecosystem-		*M. pyrifera*	N
Fish (comm. catch records)	N	**Species correlations**	
Fish (fish.-ind. survey data)	N	*M. pyrifera*/ *H. erythrogramma*	N
Correlation with SST		*U. pinnatifida*/ *H. erythrogramma*	N
C. trichoptera	N	*M. pyrifera*/ *C. rodgersii*	N
H. erythrogramma	N	*U. pinnatifida*/ *C. rodgersii*	N
C. rodgersii (location 4)	Y	*U. pinnatifida*/ *M. pyrifera*	N

2.4 DISCUSSION

In order to investigate the ecosystem effects of the banded morwong fishery, the spatial extent of the ecosystem was defined based on community composition and species abundance. Fish communities on the east coast of Tasmania were found to be significantly different to those outside this region. Previous analysis of the biodiversity survey data from 1992-1994 also defined the east coast of Tasmania as a single biogeographical region, distinct from other areas of the state (Edgar et al. 1997). The data was re-analysed in the present study to include the more recent biodiversity survey data from 1995 to 2002. The use of the commercial catch sampling data in this study did not highlight any differences in catch composition or abundance within the east coast. While the biodiversity surveys and commercial catch sampling had different selectivities due to different sampling methods, the key species identified using both methods were assumed to be representative of the inshore reef ecosystem and the banded morwong fishery. Thus, for the purposes of this study, the region could be defined as a single ecosystem, from Eddystone Point to Bruny Island.

Two of the most commonly sighted fish species, purple wrasse and blue throat wrasse, were found to be major non-target species captured in the banded morwong fishery. These species are commonly released in the banded morwong fishery due to low survival rates yet are occasionally retained for live sale. These species are also subject to a separate targeted fishery, using line fishing and trap methods. The commercial live fish wrasse fishery has had an annual production in the order of 55 tonnes from the east coast of Tasmania over the past decade (Ziegler et al. 2008). Barrett et al. (2007) found differences in the abundance of blue throat wrasse between marine reserves and fished areas, with the commercial wrasse fishery suggested to be a dominant factor behind these differences (Barrett et al. 2007). In addition to the targeted fishery, the mortality of blue throat wrasse following capture and discarding in the banded morwong fishery may be partially responsible for the reduced numbers of this species outside marine protected areas. While the survival of wrasse captured in the banded morwong fishery is thought to be low, there are few data regarding specific survival rates of discarded wrasse. As a result, there is a need to better understand the extent to which the fishery affects these species.

The investigation into key species in this chapter were of particular interest as shifts in the abundance and spatial distribution of species, due to fishing, invasive species or climate change, may indirectly impact fish production and fisheries (Hobday et al. 2005, Coll et al. 2007, Ling 2008). Changes in the distribution and production of fisheries have been observed as a result of the El Nino-Southern Oscillation and further changes are expected due to the cumulative impacts of fishing and climate change (Brander 2007). These shifts may be difficult to predict in any ecosystem due to the complexity of ecosystem structure and dynamics. As a result, ecosystem modelling is necessary to investigate the potential effects of change on the inshore reef ecosystem. The ability for change in one species to have further effects throughout ecosystems has been illustrated on numerous occasions through trophic cascades (e.g. Pinnegar et al. 2000, Shears and Babcock 2003, Worm and Myers 2003, Casini et al. 2009). In the inshore reef ecosystem, the potential for additional effects can be illustrated through the description of possible change following an increase in urchin abundance. The urchin *H. erythrogramma* has already created a number of urchin barrens on the east coast of Tasmania (Valentine and Johnson 2005). An increase in *H. erythrogramma*, in addition to an increase in *C. rodgersii* due to the strengthening of the East Australian Current and rising SST, may create additional barrens in this region. This increase in urchin abundance and grazing may have further consequences for the inshore reef ecosystem as a high level of urchin grazing has been found to reduce the abundance of native algae and aid the settlement of the invasive Japanese alga, *U. pinnatifida*, in Tasmania (Valentine and Johnson 2003, 2005). A reduction in the native algae, such as the giant kelp, *M. pyrifera*, through urchin grazing may also have other effects for the associated invertebrate and fish populations. Long-term change in the abundance of large algal species, such as *M. pyrifera*, *P. comosa* and *E. radiata*, are important as these species provide shelter and food for invertebrates (Pederson and Johnson 2006) and fish (Edgar 2000, Anderson 2001). For example, commercially important species of wrasse as well as banded morwong utilise macroalgae for shelter from predators. Shifts in the key species identified in this chapter should be investigated in detail to assess the ecosystem impact of a change in macroalgal and urchin abundance.

Species with lower abundance or catch rates, and species that have not been the subject of extensive studies, may also play a critical role in ecosystem dynamics. These little known species may become particularly important in ecosystem dynamics and fisheries as the harvest of fish continues to deplete stocks. For instance, if fishing down the food web occurs (Pauly et al. 1998), the lesser-known species from the lower trophic levels may become a valuable target for commercial fisheries. In the inshore reef ecosystem, species such as marblefish may replace the higher value, relatively well-studied banded morwong and wrasse in the export market if the capture of marblefish becomes more economically viable due to stock decline in the other species. The analyses undertaken in this chapter allowed the identification of species with high catch and sighting rates, for which little information is available. To identify species that may impact ecosystem dynamics without high catch and sighting rates, quantitative ecosystem models, such as Ecopath with Ecosim would be necessary. These models may be able to identify species or groups that are likely to influence trophic dynamics due to fishing or climate change throughout time.

The non-significance of the majority of correlations between species abundance and SST suggested any variability in species composition and abundance may have been due to alternative forces such as competition (e.g. Menge 1991, 1995, Staehr et al. 2000) and predation (e.g. Menge 1991, Shears and Babcock 2003) between species not included in the analyses. There is strong support in the literature for the influence of competition and predation, as well as environmental interactions, on ecosystem change (Hooper et al. 2005). Yet, further investigation using environmental and population models (Wielgus et al. 2007) or General Additive Models (e.g. Agenbag et al. 2003, Richardson et al. 2003, Su et al. 2008) may be useful to provide support to the findings of this study.

The a priori definition of the region and key constituents within an ecosystem can be important to ensure that species or relationships of interest are not overlooked due to the complexity of ecosystem structure (Thrush 1999). This approach also allowed ecosystem analyses to be created in context with the issue of interest (the banded morwong fishery) as well as identified a number of key fish, invertebrate and algal species. The analyses will be used in the construction of ecosystem models for the analysis of the banded morwong fishery and the inshore reef ecosystem of eastern Tasmania.

Appendix 1.1 Species sighted during the biodiversity surveys on the east coast of Tasmania.

Fish species	Invertebrate species	Algal species
Acanthaluteres spilomelanurus	Aetapcus maculatus	Acrocarpia paniculata
Acanthaluteres vittiger	Agnewia tritoniformis	Asparagopsis armata
Aetapcus maculatus	Allostichaster polyplax	Ballia callitricha
Aplodactylus arctidens	Amblypneustes ovum	Bangia spp.
Aracana aurita	Amblypneustes pachistus	Callophyllis lambertii
Aracana ornata	Aploactisoma milesii	Callophyllis rangiferinus
Arothron firmamentum	Aplysia sp.	Camontagnea oxyclada
Arripis spp.	Argobuccinium vexillum	Carpoglossum confluens
Atherinason hepsetoides	Asterias amurensis	Carpomitra costata
Atypichthys strigatus	Asterodiscides truncatus	Caulerpa annulata
Bovichtus angustifrons	Astrostole scabra	Caulerpa brownie
Brachaluteres jacksonianus	Bovichtus angustifrons	Caulerpa flexilis
Caesioperca lepidoptera	Brachaluteres jacksonianus	Caulerpa geminate
Caesioperca rasor	Cabestana spengleri	Caulerpa longifolia
Caranx dentex	Cabestana tabulata	Caulerpa scalpelliformis
Cephaloscyllium laticeps	Cenolia tasmaniae	Caulerpa simplisciuscula
Cheilodactylus nigripes	Cenolia trichoptera	Caulerpa trifaria
Cheilodactylus spectabilis	Charonia lampas rubicunda	Chaetomorpha billardierii
Chromis hypsilepis	Chlamys asperimus	Champia viridis
Conger verreauxi	Chromis hypsilepis	Codium australicum
Cristiceps australis	Conocladus australis	Codium dimorphum
Dasyatis brevicaudata	Conus anemone	Codium harveyi
Dinolestes lewini	Coscinasterias muricata	Codium pomoides
Diodon nichthemerus	Cymatium parthenopeum	Codium spp.
Dotalabrus aurantiacus	Dicathais orbita	Colpomenia peregrina
Engraulis australis	Equichlamys bifrons	Craspedocarpus ramentaceus
Enoplosus armatus	Fromia polypora	Cystophora moniliformis
Eubalichthys gunnii	Fusinus noveahollandiae	Cystophora platylobium
Forsterygion varium	Genypterus tigerinus	Cystophora retorta
Genypterus tigerinus	Gnathanacanthus goetzii	Cystophora retroflexa
Girella elevata	Goniocidaris tubaria	Cystophora subfarcinata
Girella tricuspidata	Gymnothorax prasinus	Cystophora xiphocarpa
Girella zebra	Haliotis rubra	Desmarestia ligulata
Gnathanacanthus goetzii	Heliocidaris erythrogramma	Dictymenia harveyana
Haletta semifasciata	Holopneustes inflatus	Dictyopteris muelleri
Heteroclinus johnstoni	Jasus edwardsii	Dictyota dichotoma
Heteroclinus tristis	Mesopeplum tasmanicum	Distromium spp.
Heterodontus portusjacksoni	Nectocarcinus tuberculatus	Durvillaea potatorum
Hippocampus abdominalis	Nectria ocellate	Echinothamnion hystrix
Hypoplectrodes maccullochi	Octopus sp.	Ecklonia radiata
Hyporhamphus melanochir	Octopus tetricus	Euptilota articulata
Kathetostoma laeve	Ostrea angasi	Gelidium spp.
Kyphosus sydneyanus	Paranepanthia grandis	Geniculate corralines
Latridopsis forsteri	Parascyllium ferrugineum	Gigartina crassicaulis
Latris lineata	Paratrachichthys trailli	Gigartina sp.
Lotella rhacina	Patiriella brevispina	Halicnide similans
Mendosoma allporti	Patiriella calcar	Haliptilon roseum
Meuschenia australis	Patiriella gunnii	Halophila australis
Meuschenia freycineti	Patiriella regularis	Halopteris spp.
Meuschenia hippocrepis	Penion mandarinus	Hemineura frondosa
Meuschenia scaber	Penion maxima	Homeostrichus olsenii
Myliobatis australis	Pentagonaster dubeni	Hypnea ramentacea
Nemadactylus macropterus	Petricia vernicina	Jeannerettia lobata
Neoodax balteatus	Phasianella australis	Laurencia elata
Neosebastes scorpaenoides	Phasianella ventricosa	Laurencia spp
Norfolkia clarkei	Phyllopteryx taeniolatus	Lenormandia marginata
Notolabrus fucicola	Plagusia chabrus	Lessonia corrugata
Notolabrus tetricus	Pleuroploca australasia	Lobospira bicuspidata
Odax acroptilus	Ranella australasia	Macrocystis pyrifera
Odax cyanomelas	Sassia subdistorta	Melanthalia obtusata
Omegophora armilla	Scutus antipodes	Perithalia cordata
Ophthalmolepis lineolata	Sepia apama	Peyssonelia novaehollandiae
Paratrachichthys trailli	Smilasterias multipara	Phacelocarpus alatus
Parequula melbournensis	Stichopus mollis	Phacelocarpus peperocarpus
Parma microlepis	Tasmanogobius sp.	Phyllospora comosa
Pempheris compressa	Tosia australis	Plocamium angustum
Pempheris multiradiata	Tosia magnifica	Plocamium cartilagineum

App. 1.1 (cont.)

Fish species	Invertebrate species	Algal species
Pentaceropsis recurvirostris	Trizopagurus strigimanus	Plocamium costatum
Phyllopteryx taeniolatus	Turbo undulatus	Plocamium dilatatum
Pictilabrus laticlavius	Umbraculum umbraculum	Plocamium leptophyllum
Platycephalus bassensis	Uniophora granifera	Plocamium mertensii
Pseudolabrus psittaculus	Urolophus cruciatus	Plocamium potagiatum
Pseudophycis bachus	Urolophus paucimaculatus	Polysiphonia
Pseudophycis barbata		Pterocladia capillacea
Raja whitleyi		Ptilonia australasica
Rhombosolea tapirina		Sargassum fallax
Sardinops neopilchardus		Sargassum verruculosum
Scorpaena papillosa		Sargassum vestitum
Scorpis aequipinnis		Sonderopelta coriacea
Scorpis lineolata		Sonderopelta/Peyssonelia
Sepioteuthis australis		Sporochnus spp.
Seriolella brama		Stenogramme interrupta
Siphamia cephalotes		Thamnoclonium dichotomum
Siphonognathus attenuatus		Xiphophora gladiata
Siphonognathus beddomei		Zonaria angustata
Siphonognathus tanyourus		Zonaria turneriana
Sphyraena novaehollandiae		
Tetractenos glaber		
Thamnaconus degeni		
Trachinops caudimaculatus		
Trachurus declivis		
Upeneichthys vlaminghii		
Vincentia conspersa		

3. QUALITATIVE MODELLING OF FISHERY MANAGEMENT AND TROPHIC EFFECTS IN A TEMPERATE REEF ECOSYSTEM

3.1 INTRODUCTION

Many perturbations, such as fishing, climate change and invasive species may occur in the marine environment. In order to produce an holistic investigation into the effects of such perturbations, investigations must be undertaken in an ecosystem context (Christensen and Pauly 2004). The analysis of ecosystem dynamics can be a daunting task as ecosystems may include a large number of species and environmental factors. Models of the ecosystem are used to simplify the real world to allow a greater understanding of ecosystem processes (Puccia and Levins 1985). To increase the realism of ecosystem models there has been a move towards including as much detail as possible, thereby increasing model complexity. Yet, in practice simplified ecosystem models may be necessary if data on each species and perturbation is not available. Furthermore, increased complexity in models can preclude understanding rather than aid it by complicating a general view of the system with minute detail (Fulton et al. 2003). As a result, methods of simplifying ecosystem models can be very useful.

The simplification of model structure to include the available information may be necessary in data-poor situations to allow the assessment of ecosystem dynamics. This may occur as an appropriate level of information may be lacking for the production of detailed quantitative models. Simple qualitative models (Levins 1974, 1975) may be beneficial in data-limited situations to focus research on the core problem and dynamic of interest. This can allow a greater understanding of ecosystem processes prior to undertaking more detailed quantitative studies. Qualitative models were first used in ecology by May (1973) to investigate the stability of model ecosystems and by Levins (1974) to depict, among other things, the dynamics of trophic webs. Qualitative modelling has also been used to investigate the management of the Bay Scallop fishery (Puccia and Levins 1985), trophic webs in freshwater lakes (Bodini 1998) and snowshoe hare predation (Dambacher et al. 1999). Yet, qualitative models have not been applied widely in ecology, potentially due to the fact that, in general, our scientific process is based on quantitative analyses and many researchers find moving away from empirical structure and restrictions difficult. Nonetheless, avoiding the empirical measurement of system variables, while retaining an understanding of system dynamics, can be particularly useful in natural communities that are continually changing.

Qualitative ecosystem models have fewer data requirements than quantitative models, such as Ecopath (Polovina 1984) and Multispecies Virtual Population Analysis (ICES 1994), as they use only the signs of interactions between species (+, -, 0), not their magnitude. This allows multiple types of information (e.g. measured trends or the public opinion of a resource) to be incorporated into models and may make ecosystem analyses more accessible for fisheries and ecological studies with limited quantitative data. Qualitative models may also be useful to clarify system structure and highlight where data gaps. This clarification can be used to prioritise data collection. Furthermore, qualitative models can provide valuable information on the direction of response to perturbation. Determining whether the ecosystem, as a whole, is moving in a particular direction may be more feasible than calculating specific reference points where management is initiated for individual species (Trenkel and Rochet 2003, Jennings and Dulvy 2005, Rochet et al. 2005). Such information on the direction of change, ecosystem structure and data limitations may make qualitative modelling a useful decision-making tool in the initial stages of ecosystem analysis. The benefits of qualitative models in decision-making and to aid the understanding of processes, such as fishing, were investigated in this chapter.

Qualitative models were used to investigate the dynamics of the inshore reef ecosystem of eastern Tasmania (Fig. 3.1). This region was defined as an ecosystem using biodiversity surveys and commercial catch sampling data in Chapter 2. This fishery is important for this study as its management is under review with consideration being given to the implementation of output controls in the form of quota management and a Total Allowable Catch (TAC). These limitations on catch are thought to be necessary to allow the rebuilding of banded morwong populations following a significant impact on population dynamics since the instigation of the fishery in 1994 (Ziegler et al. 2007). Being a live fish fishery with a limited domestic market, catches in the banded morwong fishery are strongly linked to market demand and fish processors encourage reduced fishing effort, particularly during periods of oversupply, when the unit price is decreased. Qualitative models were used to investigate the role of profit and demand as well as management through a TAC on the dynamics of the banded morwong fishery.

In addition to fishing pressure, the inshore reef ecosystem has been subject to an increase in the number and area of urchin barrens, areas devoid of foliose algae (Johnson et al. 2004, Valentine and Johnson 2005). Urchin barrens occur as a result of extensive grazing by urchins and commonly in temperate regions around the world, including Canada (Chapman 1981), New Zealand (Villouta et al. 2001) and Norway (Sivertsen 2006). Urchins commonly over-graze foliose algae and leave only crustose coralline algae (Rowley 1990). In Tasmania, urchin abundance has been suggested to be constrained by rock lobster (Edgar and Barrett 1999, Johnson et al. 2004, S.D. Ling, University of Tasmania, unpublished data). Further investigations found that the removal of large rock lobster, through fishing, reduced predation pressure (S.D. Ling, University of Tasmania, unpublished data), allowing urchins to increase in abundance and create barrens (Johnson et al. 2004). Yet, as the east coast of Tasmania supports a significant commercial rock lobster fishery (368t, Haddon and Gardner 2008), as well as the majority of recreational rock lobster fishing in Tasmania, the number of large individuals available to prey on urchins is relatively low. Qualitative models were produced to investigate the response of urchins to a decrease in rock lobster abundance due to fishing. In addition, the effect of barrens on associated fisheries and species within the ecosystem, such as abalone, were investigated along with the effect of reduced primary production through loss of macroalgae.

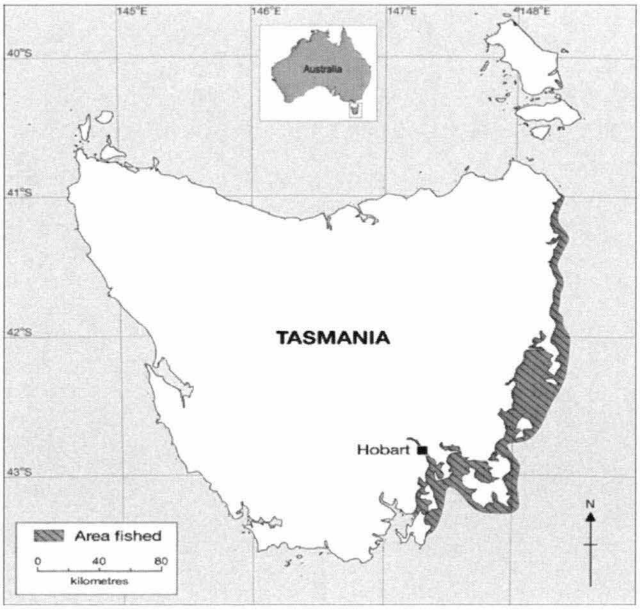


Figure 3.1 Location of the inshore reef ecosystem of eastern Tasmania, Australia, modelled in this study.

3.2 METHODS

3.2.1 Qualitative modelling and press perturbation

The growth of populations or species within an ecosystem can be described by a Lotka–Volterra system of linear equations where the per capita rate of change in abundance is controlled by rates of birth, death and migration. For example, a system of equations for a predator-prey community with three species may be described as

$$\frac{dN_1}{N_1 dt} = -\alpha_{1,1}N_1 - \alpha_{1,2}N_2 + \beta_1$$

$$\frac{dN_2}{N_2 dt} = \alpha_{2,1}N_1 - \alpha_{2,3}N_3 \tag{3.1}$$

$$\frac{dN_3}{N_3 dt} = \alpha_{3,2}N_2 - \delta_3$$

where N_1, N_2 and N_3 are species abundances for species 1 to 3, t is time, β_1 is the birth rate of species N_1 and δ_3 is the mortality rate of species N_3. Interactions between species are represented by α_{ij}, for example, $\alpha_{1,2}$ is the direct effect of species N_2 on species N_1.

The system of equations can also be represented by a graph, known as a signed digraph (Levins 1974), which displays the interactions between variables and is constructed using interaction signs (+, -, 0). For instance, the relationship between a predator (e.g. seal or fish) and its prey will be represented by a positive effect to the predator (arrow) and a negative effect (filled circle) to the prey (e.g. fish or algae) (Fig. 3.2). Density-dependent growth of a species may also be included as a negative self-effect. A path between variables may be described using a sequence of α_{ij} terms. For instance, the path from algae to fish to seals is $+\alpha_{21}\alpha_{32}$.

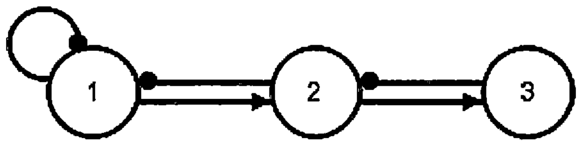


Figure 3.2 Signed digraph of a trophic relationship between algae (1), fish (2) and seals (3). Negative links are denoted by circles and positive links by arrows. A negative self-loop, as in species 1, demonstrates density-dependent growth.

Both pulse and press perturbations may affect ecosystem dynamics (Bender et al. 1984). A pulse perturbation is a short-term change in a population's abundance followed by a return to equilibrium. Press perturbations affect the ecosystem on a longer time-scale and may arise from changes to one or more of the system parameters. The community matrix **A** is composed of the direct effects between species, which are derived from the first partial derivatives of the per capita growth equations at equilibrium (Levins 1968). The inverse of the negative community matrix ($-\mathbf{A}^{-1}$) (Dambacher et al. 2002) accounts for the direct and indirect effects of press perturbations on species abundance. The inverse matrix is equal to the adjoint matrix divided by the matrix determinant

$$-\mathbf{A}^{-1} = \frac{\operatorname{adj}(-\mathbf{A})}{\det(-\mathbf{A})} \quad . \qquad (3.2)$$

As the determinant is the common denominator, predictions of the direction of response due to a press perturbation may be assessed through the adjoint of the negative community matrix (herein referred to as the adjoint matrix, Dambacher et al. 2002, 2005). The perturbed variables are read across the columns of the adjoint matrix while the response of variables to perturbation are read down the rows.

Within the adjoint matrix, predictions of response sign are calculated through the multiplication of open paths and complementary subsystems (Puccia and Levins 1985).

An open path is a series of links that starts at the perturbed variable and ends in another variable for which the response sign is in question. Links in an open path do not cross any variable twice. The associated complementary subsystem (or subsystems) includes all variables in the open path and the links between any additional variables not included in the open path. For instance, the interaction between fish and seals in Figure 3.2 has an open path between including the positive and negative links between these two variables. The complementary subsystem for this open path is the negative self-effect of algae. The product of open paths and complementary subsystems has previously been termed complementary feedback (Dambacher et al. 2002) and addends (Hosack et al. 2008); however, to avoid confusion with other types of feedback and other mathematical operations, this product will be referred to as a complementary term. The balance of the signs of these complementary terms determines the direction of change in the abundance of the variable (Dambacher et al. 2003).

Ambiguity of predictions occurs when both positive and negative complementary terms contribute to a response. If one positive and one negative term exist for a given prediction, the predicted response will be ambiguous because the probability of obtaining a positive (or negative) sign is 0.5. Ambiguity may also occur to a lesser extent. For example, four negative complementary terms and one positive term would produce a –3 in the adjoint matrix, as one negative term is cancelled by the positive term. Ambiguity in small systems can be assessed using the symbolic adjoint matrix, which identifies the terms contributing to each prediction.

The stability of a system can be assessed through the investigation of feedback. Feedback is a term used to describe the process through which an activity or change in one variable produces change in others and then returns to impact the initial variable

(Puccia and Levins 1985). For instance, a positive feedback cycle occurs when an increase (decrease) in a variable causes a further increase (decrease) to the same variable (i.e. the returning sign is unchanged). Positive feedback enhances the effect of the original change (Levins 1998). In contrast, negative feedback occurs when a decrease (increase) in a variable causes other changes that eventually result in an increase (decrease) in the same variable (i.e. the returning sign is changed). Negative feedback thereby contributes to stability in the system through the opposition of change while positive feedback contributes to instability. System stability was investigated in the fishery scenario to assess the impact of a TAC on the banded morwong fishery.

3.2.2 Models and scenarios

Two scenarios were used to investigate the inshore reef ecosystem. The first scenario examined the impact of the banded morwong fishery and included four models: a general fishery model; a demand model; and two TAC models. The second scenario examined the formation and impact of urchin barrens on the inshore reef ecosystem. Predictions of response to change and hypotheses were generated to focus data collection and guide quantitative ecosystem investigation (Chapter 6). The process of model building was included in the results for each scenario.

Fishery scenario

The first model to be produced in the fishery scenario was the general fishery model, which included catch, effort, profit and stock biomass. This model could represent the relationship between catch, effort, biomass and profit in any fishery without regulation by management. Modification of this model was then undertaken to represent the dynamics of the fishery when regulated by consumer demand (Demand model) and a TAC. The effects of a TAC were investigated (TAC model) through the inclusion of a management variable. Within these three models, the effect of profit, demand and different management regimes on the sustainability of stock biomass and the fishery were investigated.

General Fishery Model

A fishery can be treated as a consumer of a fish stock and the relationship between fishery (harvest, H) and stock biomass (B) may be depicted as a simple predator-prey relationship (Fig. 3.3).

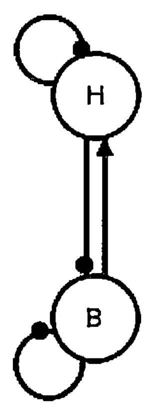


Figure 3.3 Signed digraph of harvest (H), representing a fishery, and stock biomass (B) where closed circles are negative effects and arrows are positive effects. Negative self-effects represent density-dependence or a reliance on external variables, such as market price.

The harvest variable implicitly includes catch and effort, and these aspects of the fishery may be separated out into individual variables. Catch (C) can be described as a function of effort (E), stock biomass (B) and catchability (q)

$$C = qEB \quad . \tag{3.3}$$

Price, demand and profitability also drive fishing effort because fishers reinvest their profits (P) back into boats and other fishing equipment. In the general fishery model (Fig. 3.4), effort removes fish from the ecosystem and has a direct negative effect on stock biomass. Catch is the outcome of effort (i.e. fish that were removed through effort) and therefore has no direct negative effect on the stock. Catchability is assumed to remain constant and is included implicitly in the links leading to catch from effort and biomass, as well as the link between effort and biomass. Positive feedback cycles between effort, catch and profit exist in this model (Fig. 3.4). This model does not include output controls, such as a TAC or total allowable effort (TAE), or input controls placed on the fishery by a management body.

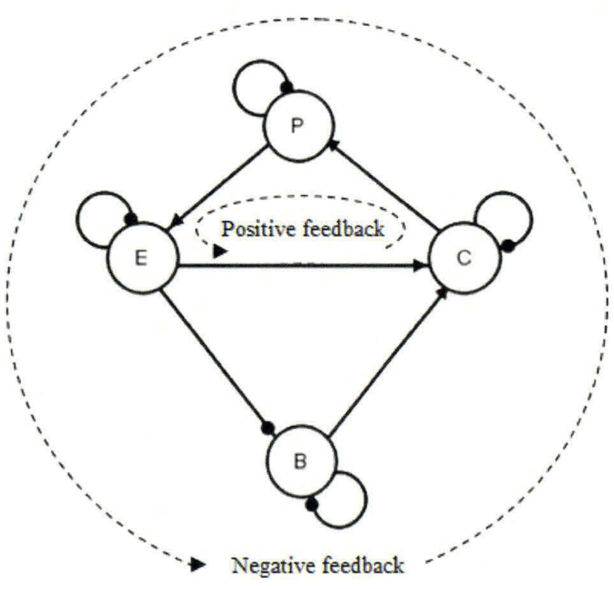


Figure 3.4 Signed digraph of the general fishery system where B is biomass, C is catch, P is fisher profit and E is effort. Dashed lines show the locations of positive and negative feedback cycles. Closed circles represent negative effects while arrows represent positive effects.

Demand model

An increase in catch can saturate the market, thereby lowering the demand for resources and eventually decreasing fisher profits. A demand (D) variable was added to the general fishery model to represent the limitation of profit when demand for the product is reduced (Fig. 3.5b). This model is a stepping-stone between the general fishery model and the TAC model. The adjoint matrix for this model was not included in these results.

The negative effect linking demand and catch usually only occurs in the fishery in summer, when catch rates are high, and a glut of fish saturates the market. As a result, fish processors may encourage banded morwong fishers to decrease their effort to allow price and profits to increase. In winter, banded morwong are more difficult to catch and demand is not always met by supply. As a result of the seasonal differences in the effect of demand, both the positive link between catch and profit and the links between catch, demand and profit were retained in order to represent the fishery throughout the year. A negative effect links catch and demand because the demand for fish products is reduced as catch increases.

TAC model

A TAC has been represented in this model as a limitation of both catch and effort by management (Fig. 3.5c). This is because of the relationship between catch, effort and biomass in the catch equation (Eq. 3.3), which determines that an independent change in effort or biomass will result in a change in catch. In addition, restricting both catch and effort should occur simultaneously because, once the TAC is reached, management restricts all effort in the fishery.

TACs are designed to be at a sustainable catch level and are determined by management using information on population dynamics, as well as log book and catch rate data. The use of this information was included through positive links from catch and effort to management. In addition, fishery-independent stock assessments are undertaken to determine whether the stock biomass is sufficient to sustain catches. This information was represented using a negative link from biomass to management as an increase in biomass may reduce the need for management to act. In general, management regulations, such as TACs, are reassessed periodically to ensure they remain at an appropriate level to ensure the sustainability of stocks. The potential for increased regulation in response to catch and effort levels has been shown through negative links from management to catch and effort.

The fishery, represented here by catch and effort, is essentially a predator with a type 1 functional response (Holling 1959, Dambacher and Ramos-Jiliberto 2007), where the standard consumption rate of a predator increases linearly until a threshold is attained. After this threshold the consumption rate remains constant, even with increasing prey biomass. Due to the existence of a threshold effect, once the TAC is reached a second model was necessary to describe the banded morwong fishery. In this model, the link between profit and effort was removed because no additional fish can be sold (Fig. 3.5d). This removes the incentive for fishing to continue legally and all catch and effort for this species is halted until the following year. The response predictions following change in the fishery or management were not investigated in this model as the fishery does not exist at this point in time.

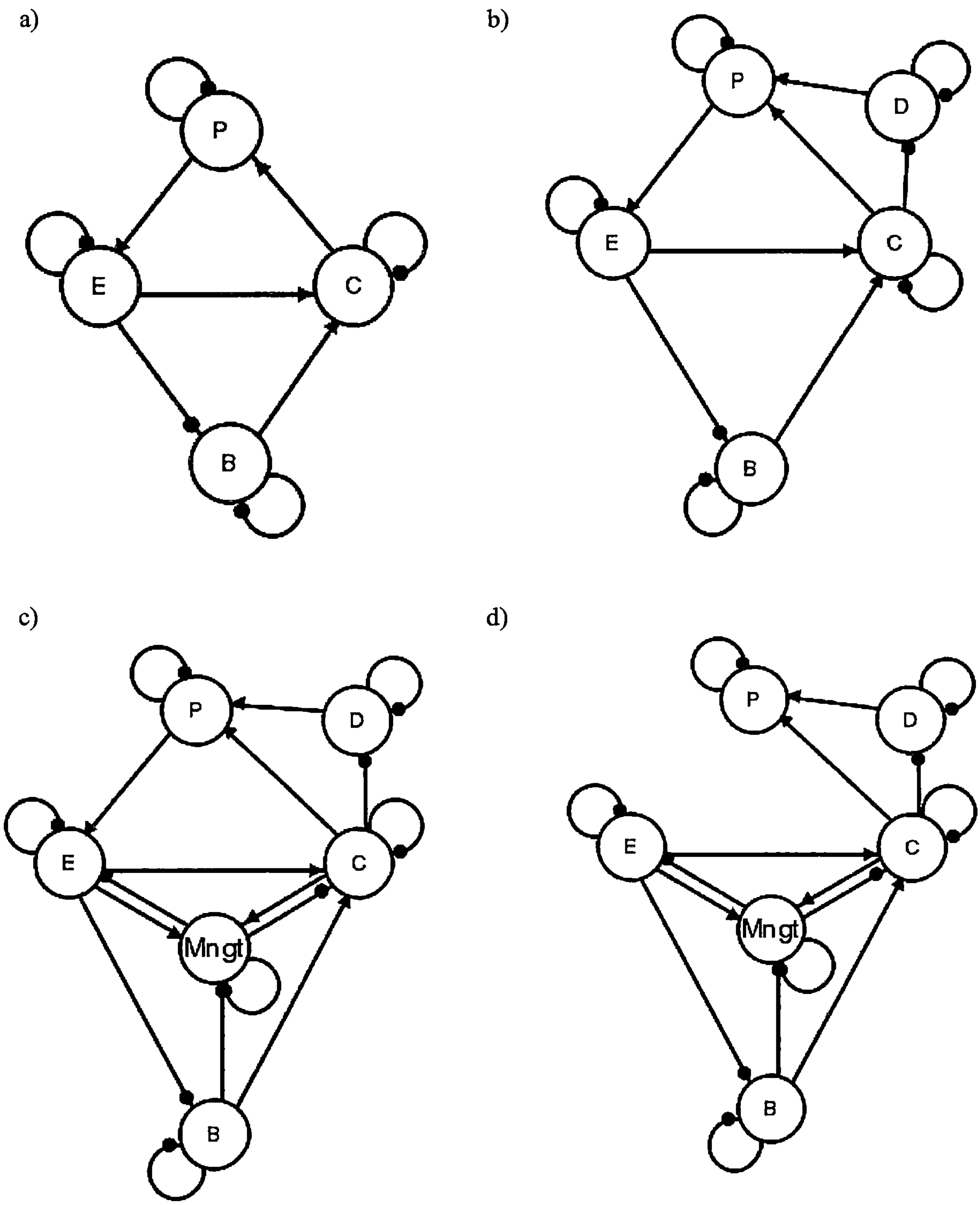


Figure 3.5 Progression of the Fishery scenario models from the General fishery model (a) to the Demand model (b) and the TAC models, before (c) and after the TAC is reached (d). B is biomass, C is catch, P is profit, E is effort, D is demand and Mngt is management.

Urchin barren scenario

Urchins graze foliose algae preferentially over crustose algae (Wright et al. 2005) and, at high urchin abundance, this may result in the formation of urchin barrens. Urchin barrens may be formed when the equilibrium abundance of foliose algae is near zero. Near zero equilibrium density can occur when the predatory interaction between urchins and foliose algae is strong. At this equilibrium abundance, foliose algae may only exist in small patches (i.e. there is localised extinction). Urchins are then forced to switch prey resources and feed on crustose algae, as in urchin barrens. Barren formation can be shown in a set of simple predator-prey models (Fig. 3.6) where a positive press perturbation to urchins increases grazing on their preferred prey, foliose algae, and leaves crustose algae as the sole resource in the system.

Rock lobster were included in the system (Fig. 3.6) as they have been suggested to constrain urchin abundance in Tasmanian marine reserves (Edgar and Barrett 1999). This occurs by altering the abundance and death rate of urchins (Johnson et al. 2004, S.D. Ling, University of Tasmania, unpublished data) and may allow the foliose algal equilibrium abundance (N^*) to increase substantially above zero, thereby reducing the potential for the formation of urchin barrens.

The model was then incorporated into a larger model to investigate the impacts of urchin barren formation and decreased algal abundance on the community (Fig. 3.7). Abalone and rock lobster were included as separate variables due to their commercial significance while a 'fish' variable was used to represent banded morwong and the common byproduct species caught in the banded morwong fishery, i.e. wrasse (*Notolabrus* spp.), long-snouted boarfish (*Pentaceropsis recurvirostris*) and bastard trumpeter (*Latridopsis forsteri*). Decreasing algal abundance (or standing crop) may occur as a result of urchin grazing or increased phytoplankton abundance and shading (Short and Burdick 1996). In order to investigate both potential causes of reduced algal abundance, separate inputs to urchins and macroalgae were undertaken. In contrast, rock lobster fishing and increasing urchin abundance are both currently occurring in Tasmania, as a result, a simultaneous negative perturbation to rock lobster and a positive perturbation to urchins was undertaken.

The removal of large rock lobster through fishing has been proposed to release urchins from predation and allow urchin populations to increase in abundance (S.D. Ling, University of Tasmania, unpublished data). In contrast, smaller rock lobster, including small individuals within the legal size range, can prey upon urchins but not to the extent that urchin abundance and grazing is limited. The first model (Fig. 3.7a) represents the system where large rock lobster are present while the second model (Fig. 3.7b) represents the system when larger individuals have been fished and are no longer present to decrease the abundance and grazing of urchins. Two separate models were used to represent different sized rock lobster instead of using two life-stages in one model (sensu Dambacher et al. 2005) because fishing was assumed to remove the majority of large rock lobster from the system. Small rock lobster were assumed to be removed from the system as soon as they reached the legal size limit. The model with larger individuals therefore represents a system with a higher size limit. A positive effect was applied to show the use of foliose algae by small wrasse for shelter. As in Figure 3.3, fisheries were represented by a single harvest variable.

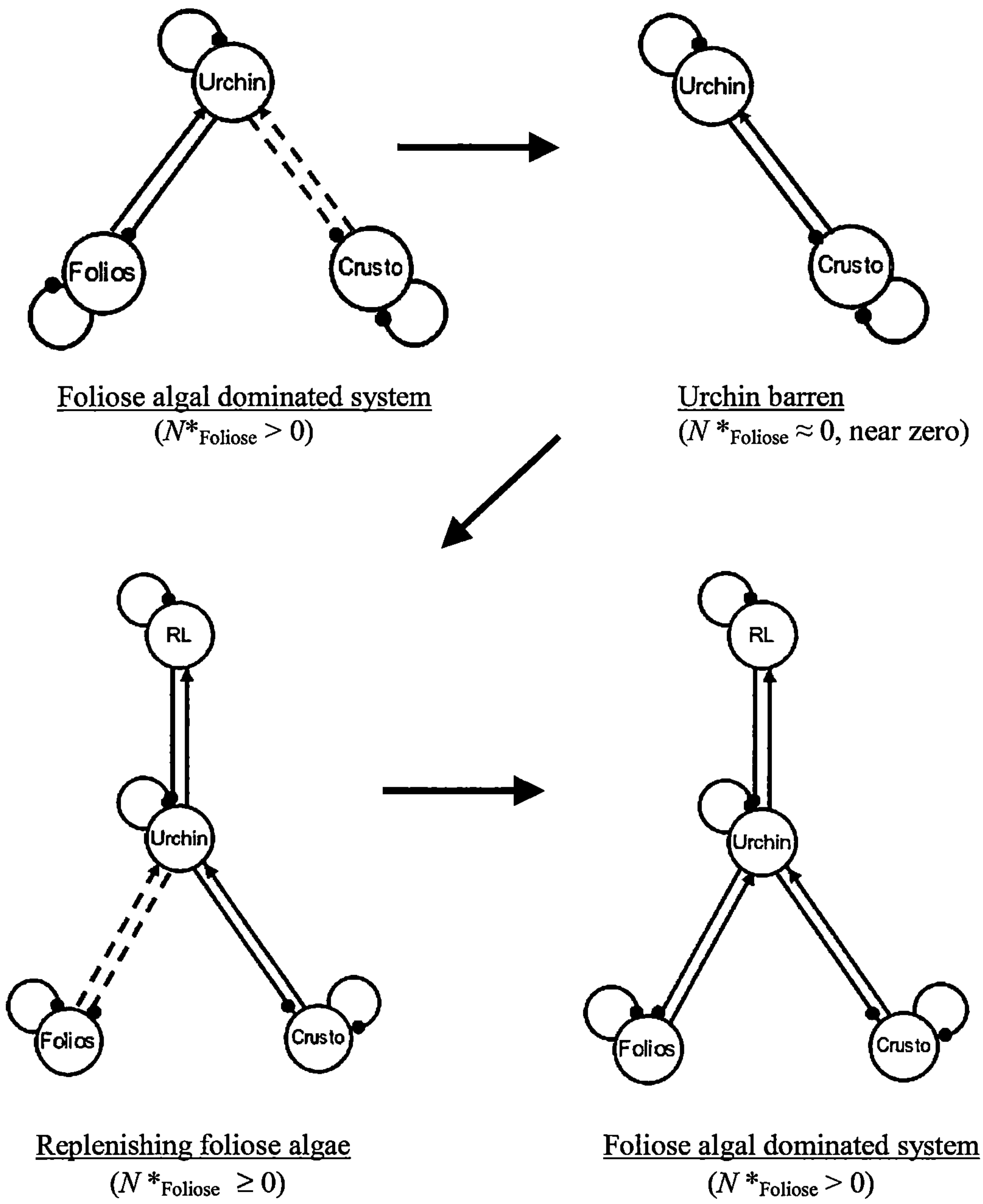


Figure 3.6 An increase in urchin abundance in a foliose algal dominated system may shift the system into an alternate state (urchin barren). This occurs as urchins preferentially feed on foliose algae (strong interaction, full lines) over crustose algae (weak interaction, dashed lines). Rock lobster (RL) predation on urchins may allow the system to return to a foliose algal dominated system by reducing the interaction between urchins and foliose algae. Closed circles represent negative effects and arrows represent positive effects. Abbreviations are crustose algae (Crusto) and foliose algae (Folios). $N^*_{Foliose}$ is the equilibrium abundance of foliose algae.

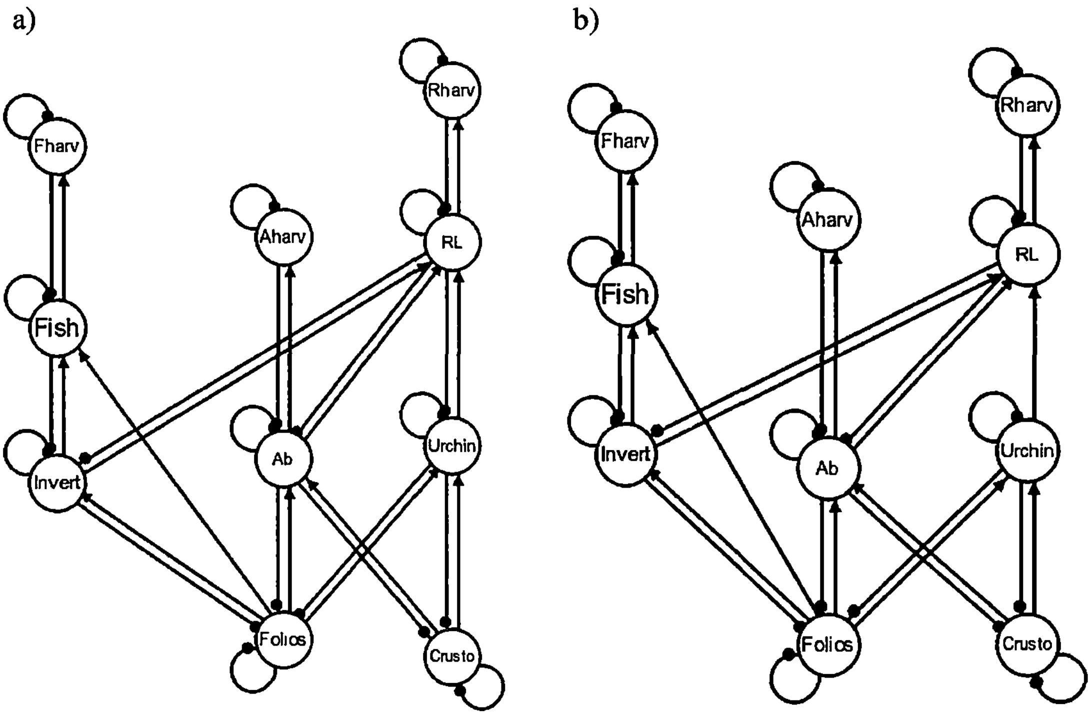


Figure 3.7 Signed digraph representing the relationship between urchins, foliose algae (Folios), crustose algae (Crusto) and rock lobster (RL) and the three main types of fishing in Tasmanian coastal waters: scalefish (Fharv); abalone (Aharv); and rock lobster (Rharv). An abalone (Ab) variable and a fish variable (Fish) comprising banded morwong, bastard trumpeter, blue throat wrasse and purple wrasse were also included. The invertebrate variable (Invert) represents the prey of these fish species, such as gastropods. A positive effect was employed to show the use of foliose algae by small wrasse for shelter. Model a) represents the system with large rock lobster while model b) represents the system when large rock lobster have been removed by fishing.

3.3 RESULTS

3.3.1 Fishery scenario

General fishery model

Positive feedback cycles between effort, catch and profit contribute to instability in this model and arise from the over-capitalisation of the fishery. Alternative paths from profit to catch create ambiguity in the predicted response of catch to perturbation. Increased profit was predicted to either increase or decrease catch depending on the strength of the links between effort and biomass, and effort and catch. The strength of these links is important as the model predicts an increase in profit always enhances effort (when output controls are not used by management) and this may result in the over-capitalisation of the fishery. Overfishing may occur when the feedback cycle P_E_C_P (positive feedback cycle, Fig. 3.4), driven by the desire to enhance profits, is strong. The reinvestment of these profits back into effort can continue to fuel harvest without reference to changes in stock biomass. In contrast, the fishery will be regulated and the potential for overfishing reduced if the feedback cycle P_E_B_C_P (negative feedback cycle, Fig. 3.4), which is regulated by biomass, is strong.

The predicted response of stock biomass to an input to itself is also ambiguous (Table 3.1). This occurred as both positive and negative complementary terms contributed to the calculation of prediction sign (Fig. 3.5). Knowledge of the strength of the links in each term is then necessary to determine whether the predicted response will be positive or negative. For example, if the self-regulation of catch, profit and effort (Fig. 3.5a) is stronger than the positive cycle between these three variables (Fig. 3.5b), the predicted response would be positive and the stock biomass would be expected to increase in abundance. In other words, the regulation of catch, profit and effort must constrain the continual drive to increase profits. If this does not occur, the stock will be unable to increase to sustainable levels following harvest and the system would collapse.

When an input to biomass is predicted to negatively impact itself, the population will be unsustainable at the current fishing level. In contrast, when an input to biomass results in a positive prediction, the population can sustain itself at current fishing levels.

Table 3.1 Adjoint of the negative community matrix (-A) displaying response predictions to perturbations or increases in each variable in the General Fishery Model (Fig. 3.4). '+' increase, '-' decrease, '+,-' ambiguous response.

Predictions/Perturbations	*B*	*C*	*E*	*P*
B	+,-	-	-	-
C	+	+	+,-	+,-
E	+	+	+	+
P	+	+	+,-	+

a) Sign of complementary term = +

b) Sign of complementary term = -

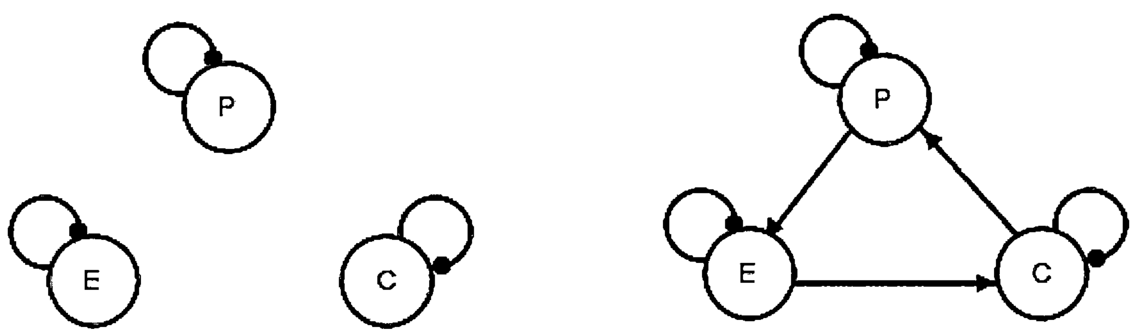


Figure 3.8 Complementary terms used in the calculation of prediction sign for the impact of a change in stock biomass on itself. Complementary term a) where catch (C), profit (P) and effort (E) are self-regulated is positive, while term b) with a positive cycle between catch, profit and effort, is negative.

TAC model

The application of a TAC by management reduced the potential for the positive feedback cycle between catch, profit and effort to drive the system to collapse. In the general fishery model, an increase in profit was predicted to either increase or decrease catch. This occurred as a result of the two countervailing feedback cycles shown in Figure 3.4. In addition to the two feedback cycles identified in this model (one positive, one negative), another two negative feedback cycles contributed to a predicted negative response of catch to profit in the TAC model (Table 3.2). These cycles were P_E_B_Mngt_C_P and P_E_Mngt_C_P, with the regulation of catch by management an important aspect of both cycles. These negative feedback cycles increase the stability of the system. The role of management is also important as it was predicted to increase stock biomass by reducing effort and catch through the paths: Mngt_E_B and Mngt_C_P_E_B. If the TAC was set at a sustainable level, these management restrictions should allow the stock biomass to rebuild prior to the next fishing season.

Table 3.2 Adjoint of the negative community matrix (-**A**) displaying response predictions to perturbations or increases in each variable in the TAC Model (Fig. 3.7). '+' increase, '-' decrease, '+,-' ambiguous response.

Predictions/Perturbations	*B*	*C*	*E*	*P*	*Mngt*	*D*
B	+	+	-	-	+	-
C	+	+	-	-	-	-
E	+,-	-	+	+	-	+
P	+,-	+,-	+,-	+	+,-	+
Mngt	-	-	+	+	+	+
D	+,-	+	+	+	+	+

General fishery model- summary of results

Two predictions were obtained in the four models in the fishery scenario that require further investigation. The first was that over-capitalisation of fisheries can cause instability in the fishery system. The second was that a TAC increases the stability of fishery systems by allowing stock biomass and, eventually, catch to increase.

3.3.2 Urchin barren scenario

Following an increase in the abundance of foliose algae, the importance of this algal variable was observed in both models in Figure 3.7. Positive responses were predicted in all variables, excluding crustose algae. Similarly, a decrease in foliose algae was predicted to have a negative impact on all variables excluding crustose algae.

An increase in urchins through a positive press perturbation produced consistent results between models with and without large rock lobster. Negative responses were predicted to occur in the scalefish and abalone fisheries, as well as decreases in the abundance of invertebrates, foliose and crustose algae. In both models, rock lobster fisheries were predicted to be positively affected by an increase in urchin abundance. This occurred as a result of the predatory relationship between urchins and rock lobster. A negative press perturbation, which reduces rock lobster fishing (signs in Table 3.3 are reversed for a negative press perturbation), would therefore be predicted to positively effect foliose algae. This may occur because rock lobster can increase predation on urchins thereby releasing algae from predation.

Different results between models with and without large rock lobster were observed in urchins, fish, the scalefish and abalone fisheries, and crustose algae following an increase in rock lobster (Table 3.3). An increase in large rock lobster, as may occur if the size limit was increased, reduced the abundance of urchins and positively affected fish, crustose algae and the scalefish fishery. In contrast, an increase in small rock lobster increased the abundance of urchins and negatively affected fish, crustose algae and the scalefish fishery. These results occurred as a result of the lack of predatory pressure on urchins by small rock lobster. In addition, different predictions were observed in the same variables following an increase in the rock lobster fishery (circles, Table 3.3).

Table 3.3 Adjoint of the negative community matrix (-**A**) displaying response predictions to perturbations or increases in each variable in the Urchin Barren Model with (Fig. 3.7a) and (including the effects of fishing) without large rock lobster (Fig. 3.7b). '+' increase, '-' decrease, '+,-' ambiguous response.

Pred./ Pert.	*Foliose algae*	*Urchins*	*Rock lobster*	*Fish*	*Abalone*	*Fish harvest*	*Abalone harvest*	*RL harvest*	*Crust. algae*	*Invert.*
With large rock lobster (Fig. 3.10a)										
Foliose algae	+	-	+	+	-	+	-	-	-	-
Urchins	+	+	-	+	-	+	-	+	+	-
Rock lobster	+	+	+	-	+	-	+	-	+	+
Fish	+	-	+	+	-	+	-	-	-	+
Abalone	+	-	-	+	+	-	-	+	+	-
Fish harvest	+	-	-	+	+	+	-	+	+	-
Abalone harvest	+	-	+	+	-	+	+	-	-	+
RL harvest	+	+	+	-	+	-	+	+	+	+
Crust. algae	-	-	+	-	-	+	+	-	+	+
Invert.	+	-	-	-	-	+	+	+	-	+
Without large rock lobster (Fig. 3.10b)										
Foliose algae	+	-	+	+	-	+	-	-	-	-
Urchins	+	+	+	+	-	+	-	-	+	-
Rock lobster	+	+	+	-	+	-	+	-	+	+
Fish	+	-	-	+	-	+	-	+	-	+
Abalone	+	-	-	+	+	-	-	+	+	-
Fish harvest	+	-	-	+	+	+	-	+	+	-
Abalone harvest	+	-	-	+	-	+	+	+	-	+
RL harvest	+	+	+	-	+	-	+	+	+	+
Crust. algae	-	-	-	-	-	+	+	+	+	+
Invert.	+	-	-	-	-	+	+	+	-	+

A simultaneous positive impact to the rock lobster fishery and urchins was investigated as this would be expected if fishing and increasing urchin abundance continued to occur. Regardless of the size of rock lobster, this perturbation resulted in negative effects to all variables in the system excluding urchins, invertebrates and the rock lobster fishery.

Urchin barren model- summary of results

The first prediction generated from the TAC models was that large rock lobster could reduce urchin abundance and positively impact all variables excluding invertebrates. Secondly, an increase in urchins may negatively affect the scalefish and abalone fisheries while positively affecting the rock lobster fishery. Finally, these models suggest a decrease in foliose algae would result in decreases to all ecosystem members and fisheries, excluding crustose algae.

3.4 DISCUSSION

Qualitative models and predictions were produced to represent banded morwong fishing impacts and urchin barrens on the inshore reef ecosystem of eastern Tasmania. These models were produced to generate predictions to guide and focus further research and to increase the understanding of ecosystem dynamics. The fishery models identified the need to further investigate the impacts of overcapitalisation and the use of a banded morwong TAC in the inshore reef ecosystem. These models supported the suggestion that unregulated fisheries have the capacity to cause stock collapse and destabilise fishery systems (Rosenberg et al. 1993). This was seen to be the case in the general fishery model where, if the fishery was driven largely by profits (overcapitalisation) through positive feedback between catch, profit and effort, an increase in biomass spurred on further effort and decreased the stock biomass in the system. Two examples of overcapitalised 'fisheries' are whaling (Clark 1981) and the Newfoundland Cod fishery (Roughgarden and Smith 1996), where harvesting continued to occur despite low numbers. In contrast, in the TAC model, if catch, profit and effort were regulated and were at a sustainable level, an increase in biomass was predicted to further increase biomass. This would determine that over-harvest by fisheries would not occur. Once the TAC has been reached, fish can no longer be sold legally and profits are not an incentive to continue fishing. Incorporating a TAC into the fishery model increased its stability by reducing the chance of profits driving the fishery to collapse. Nonetheless, the use of a TAC would be ineffective if it is set at an unsustainable level. This model assumes management is strong enough to combat any increase in demand that may fuel profits and increased harvest through illegal fishing. In practice, unsustainable TACs may be set by management due to social and economic pressure (Fujita et al. 1998) or uncertainty in the status of resources (Rosenberg et al. 1993). If an unsustainable TAC was implemented in the banded morwong fishery, catch and effort could continue to increase until the fishery or population collapsed.

The urchin barren scenario highlighted the importance of the relationship between foliose algae, urchins and rock lobster in the ecosystem. The detailed urchin barren models (with and without large rock lobster) predicted an increase in large rock lobster would decrease urchin abundance, potentially reducing the likelihood of the collapse of foliose algal populations. The ability for species abundances to change dramatically as a result of a positive press perturbation on predators, rock lobster in this case study, is not a new concept (Pimm 1979, Bender et al. 1984). Yet, the use of near zero equilibrium abundance in the qualitative models demonstrates a basic mechanism through which the foliose algal-dominated and urchin barren states may exist in Tasmanian waters. The findings of the qualitative models supported earlier work regarding rock lobster predation in Tasmania (Edgar and Barrett 1999, Johnson et al. 2004, S.D. Ling, University of Tasmania, unpublished data). The models of rock lobster predation, with and without large individuals, were based on these earlier field studies and found a reduction in urchins due to predation by large rock lobster was predicted to allow foliose algae to increase in abundance. An increase in some macroalgal species was observed at Maria Island on the east coast of Tasmania following protection from fishing. Yet, this change could not be attributed to an increase in lobster predation on urchins alone (Edgar and Barrett 1999). Additional field investigations are necessary to clarify the impact of rock lobster predation on urchins on the abundance of foliose algae within the ecosystem.

The results of the qualitative models are useful for ecosystem analyses as they include a number of additional variables (i.e. scalefish fishery, abalone fishery, abalone, invertebrates and fish) that were not included in previous studies of urchin barrens in Tasmania. This provided an opportunity for alternative predictions to arise. Investigation into the indirect effects of large rock lobster highlighted a predicted increase in scalefish abundance, as well as a decline in urchin abundance. In contrast, when only small rock lobster were present, decreased abundances of fish (banded morwong, bastard trumpeter, blue throat wrasse and purple wrasse) were predicted to occur, negatively impacting the fishery. This highlights the importance of predation and supports suggestions that a reduction in the fishing of large rock lobster could ameliorate the effects of overgrazing by urchins (S.D. Ling, University of Tasmania, unpublished data).

Similarly to species interactions, environmental changes may have a significant impact on ecosystem structure and dynamics. Increasing SST and altered coastal circulation patterns, as a result of climate change, have the potential to negatively impact macroalgae in Tasmanian waters (Hobday et al. 2006). Alterations in the primary production of many systems can cause significant changes to occur up the food chain (e.g. Carpenter et al. 1985, Pace et al. 1999). The results of a negative press perturbation to foliose algae, which caused reductions in all higher trophic variables, were therefore not surprising. Further investigation into these potential changes using quantitative analyses is warranted, as the simultaneous impacts of urchins and their predators may influence the speed at which changes occur within the ecosystem.

The qualitative models produced in this study were useful in the investigation of the structure, dynamics and stability of the inshore reef ecosystem with regard to fishery impacts, the use of a TAC by management and overgrazing by urchin. The predictions and hypotheses generated during qualitative modelling have been used to guide and focus data collection and further analyses using quantitative ecosystem models. Specifically, the stability of the fishery and banded morwong population will be assessed through the simulation of a TAC using Ecopath with Ecosim (Chapter 6). The response of all commercial target and byproduct species, catch biomass and urchin abundance to the fishery under a TAC shall also be investigated during this simulation. A competitive TAC was incorporated into banded morwong fishery management in October 2008, with the TAC set at status quo (i.e. equal with recent catch levels). The utility of a TAC set at this level for sustainability will also be investigated using simulations. Ecosim will be used to investigate the potential for further expansion of urchin barrens and the overall impact urchins may have on the ecosystem. Alterations to current rock lobster fishing rates and urchin abundance will be used to investigate hypotheses regarding the formation of urchin barrens. Finally, simulations investigating a reduction in primary production as may occur due to climate change will be undertaken. The models in this chapter present an important framework with which to guide and focus further ecosystem investigation.

4. SURVIVAL OF RETAINED AND DISCARDED FISH FROM GILLNETS IN A COMMERCIAL TASMANIAN LIVE FISH FISHERY

4.1 INTRODUCTION

Knowledge of total fishing induced mortality is necessary to accurately assess stock sustainability (Parker et al. 2003) and produce realistic models for the prediction of fishing effects. Total fishing mortality is composed of both reported and unreported catch (e.g. discarded non-target species) (Broadhurst et al 2006) and is particularly important in fisheries that use relatively unselective fishing gear, such as gillnets. Discarding involves the return of unwanted fish (both dead and alive) to the ocean following capture (Goni 1998) and is commonplace in these fisheries as a wide range of target and non-target species may be captured (Singh and Weninger 2009). Post-release mortality of discarded fish may occur as a result of injuries obtained during capture (Bettoli and Scholten 2006, Rudershausen et al. 2007) or as a result of the predation of weak or disorientated fish upon release (Hill and Wassenberg 2000). The mortality of these discarded fish will therefore contribute to the total fishing induced mortality. Discarded fish are generally non-target species with little or no commercial value; however, target species may also be discarded due to size or catch limits. High-grading, based on factors such as fish condition or market preference for size, may also play a role in discarding.

Discard mortality does not always occur immediately and is difficult to quantify, yet should be included in investigations of fishery effects (Davis 2002). In many fisheries, specific records of the species, number and condition of discards are not collected and substantial uncertainty exists in the estimation of discard and survival rates

(Alverson et al. 1994, Ross and Hokenson 1997, Aarts and Poos 2009). This is an issue that needs to be addressed to accurately estimate the effects of fishing on target and non-target species within the broader ecosystem.

In the live fish trade, mortality following capture results in resource wastage and can have a detrimental economic impact on the fishery (Hall et al. 2000). This is because injury or death due to capture and handling determines the fish may not be suitable for sale or attract a reduced value. There are two prominent live fish fisheries in Australia, one targeting tropical species, principally coral trout (*Plectropomus leopardus*, F. Serranidae) (Rimmer and Franklin 1997), and another targeting the temperate species, banded morwong (*Cheilodactylus spectabilis*, F. Cheilodactylidae) and wrasse (*Notolabrus* spp., F. Labridae). Despite potentially reduced fishery profits, factors influencing the survival rates of these species following capture are not well understood. Two studies estimated the rate of survival in retained coral trout prior to sale (Rimmer and Franklin 1997, Frisch and Anderson 2000). In addition, the study by Rimmer and Franklin (1997) investigated factors that increased the survival of wrasse and banded morwong in the live fish trade with regard to transport techniques and factors such as tank type and light levels. The role of capture techniques, including soak time of nets and capture depth, on the survival of retained and discarded individuals has not been formally investigated in the banded morwong live fish fishery.

Although a total of 45 non-target species were taken during commercial catch sampling in the banded morwong fishery (1993-2005), only six of these species were caught in relatively high numbers (Chapter 2). The most commonly caught non-target species of commercial value (byproduct species) were: bastard trumpeter (*Latridopsis forsteri*, F. Latrididae); blue throat wrasse (*Notolabrus tetricus*); purple wrasse (*N.*

fucicola); and long-snouted boarfish (*Pentaceropsis recurvirostris*, F. Pentacerotidae) (Chapter 2). The retention of these species is restricted by size limits in Tasmania and individuals outside of the legal size range are assumed to be discarded. Based on the size composition of catches taken as part of commercial catch sampling, approximately 18% of bastard trumpeter (minimum size limit of 350 mm total length (TL)), 16% of blue throat wrasse (legal minimum size limit of 300 mm TL), 48% of purple wrasse (minimum size limit of 300 mm TL) and 80% of long-snouted boarfish (minimum size limit of 450 mm TL) captured by the fishery were assumed to be discarded. In contrast, non-target species without commercial value (bycatch species) were typically not retained and constituted approximately 45% of the total catch by numbers during commercial catch sampling (Chapter 2). The survival rates of these non-target species, once discarded, are also unknown. The two most commonly caught bycatch species are marblefish (*Aplodactylus arctidens*, F. Aplodactylidae) and draughtboard shark (*Cephalloscyllium laticeps*, F. Scyliorhinidae) (Chapter 2). Little is known about the impact of the fishery on these target and non-target species and, as a consequence, the impact of the banded morwong fishery on the inshore reef ecosystem.

This study investigated the short-term survival and factors influencing survival in the seven most commonly captured species in the banded morwong fishery. The study was divided into two components. First, fish condition following commercial capture was investigated in the field. Second, aquaria trials were undertaken to assess the relationship between condition and short-term survival. These data were used to test the hypothesis that condition may be used as a proxy for survival potential (Rudershausen et al. 2007), thereby enabling the effects of the commercial fishery on non-target species to be assessed. These data were used to estimate the overall fishing mortality associated with the banded morwong fishery in Tasmania and were incorporated into quantitative ecosystem models to investigate total fishery impacts (Chapter 6).

4.2 METHODS

4.2.1 Fish condition

To assess fishery effects, the condition of fish following gillnet capture was observed on 25 fishing days between April 2005 and July 2006 during commercial catch sampling and research fishing involving standard commercial gillnet practices. Sampling was opportunistic in order to coordinate with fishers and their selected fishing locations. Gillnets were set for between 20 minutes and 6.5 hours, with soak times determined by the fisher and varying as a result of weather and catch rates. The presence of seals usually resulted in earlier gillnet retrieval as they commonly remove fish from the nets and damage the gear. Commercial mesh sizes were used which range between 130 and 140mm stretched mesh, and nets were set in depths between 5 and 22m, with a mean of 9m. Observations were undertaken at 20 locations on the east coast of Tasmania (Fig. 4.1) with a total of 4181 individuals examined from the seven study species.

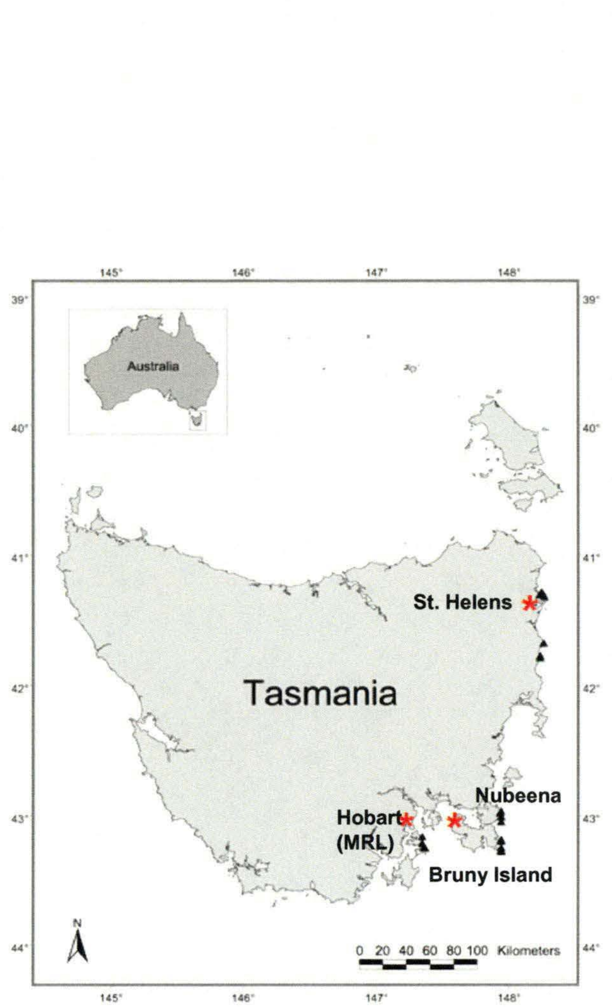
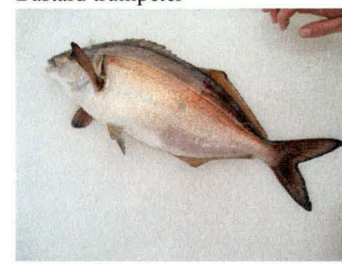
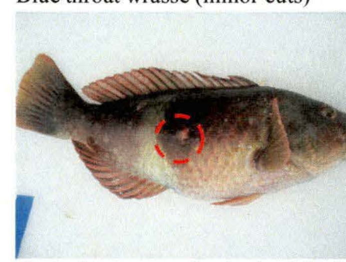
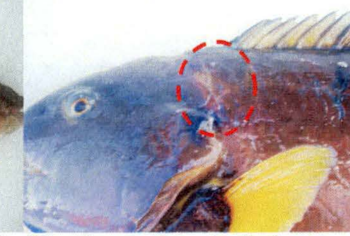
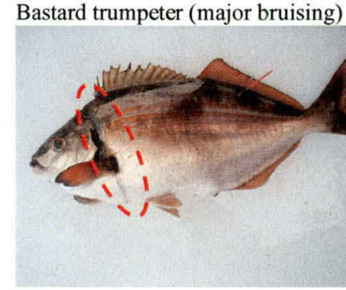


Figure 4.1 Map of Tasmania showing fish collection and condition assessment locations (n = 20, triangles). Aquaria locations used in the survival experiments (see Survival experiments) are also shown (n = 3, stars).

The condition of each fish was assessed upon removal from gillnets and classified into one of four grades according to the nature and extent of injuries. Condition grades ranged from healthy, with little or no obvious external damage (1), to dead at capture (4). Fish condition was rated according to factors such as the extent of bleeding, scale loss, barotrauma and spine damage (Table 4.1). Live fish with some evidence of damage as a result of capture were classed as Condition 2 or 3. Condition 2 fish exhibited minor physical damage to less than 5% of the body. In contrast, Condition 3 fish exhibited major physical damage to more than 5% of the body. All fish were measured to enable the potential effect of size on condition to be investigated.

Generalised Linear Modelling (GLM) with backward stepwise regression was used to investigate the importance of season, fish length, species, soak time and fishing depth on fish condition. Interactions between significant factors (species × season) were also investigated using this method. Analysis of variance and post-hoc Bonferroni pairwise comparisons were then used to investigate the specific cause of any significant factors. All statistical analyses were conducted using Systat 9.

Table 4.1 Description of condition grades used to assess fish captured off the east coast of Tasmania in 2005-2006. Fish may have one or more injuries and if multiple injury categories were present fish were placed in the poorest condition according to the condition descriptions.

Condition	Description	Example photos	
1	No obvious external damage, healthy and lively: • no cuts; • no bleeding; • no gill damage; • no scale loss; • no bruising; • no barotrauma.	Bastard trumpeter 	Purple wrasse 
2	Minor physical damage, lively: • small amount of bruising (in one area only, < 5 % of body); • minor cuts (superficial cuts, <5% of body); • small amount of bleeding from superficial cuts; • minor scale loss (<5% of body); • minor barotrauma effects.	Blue throat wrasse (minor cuts) 	Blue throat wrasse (minor scale loss) 
3	Major physical damage, sluggish: • major dorsal or pectoral spine damage (broken spines, spine covering removed); • extreme barotrauma (including protruding gut); • large cuts (>5% of body); • large amount of bruising (multiple areas, >5% of body); • gill damage (bleeding); • scale loss (>5% of body); • heavy bleeding.	Bastard trumpeter (major bruising) 	Blue throat wrasse (major barotrauma) 
4	Dead at capture		

4.2.2 Aquaria trials

The same areas and fishing methods utilised in the *Fish condition* component of the study were used to collect a total of 224 fish for aquaria trials (Fig. 4.1). Two commercial operators captured fish for use in these experiments, in addition to fish captured by researchers. Fish collection occurred in early spring (n=94) and late summer (n=130) due to the results of section 4.2.1.

Fishing was undertaken in close proximity to the location of aquaria at: the Marine Research Laboratories (MRL) in Hobart; the holding tanks of a commercial fisher based at Nubeena on the Tasman Peninsula; and a fish processing facility in St. Helens on the north-east coast of Tasmania (Fig. 4.1). Different locations were used to reduce travel times between fishing areas and holding facilities with the aim of reducing the stress placed on the fish.

After capture, the condition of each individual was assessed and the fish tagged with a t-bar tag for identification. T-bar tags were used as they were not found to impact the survival of the study species (S.J. Metcalf, unpublished data). Fish were placed in 300 and 1000L onboard holding tanks following capture. The size of the tank used depended on the fishing boat in use. All tanks were frequently flushed with fresh seawater and the maximum number of fish carried at one time was 20 per boat (in one tank). Fish were retained in onboard tanks for between one and five hours and remained in these tanks during transport to aquaria. Road transport times ranged between 25 and 45 minutes. Any changes to condition during holding and transport, including mortalities, were recorded. Fish that died or were in poorer condition following transport remained in the condition grade assigned at capture. For example, a fish that was determined to be in condition 2 at capture but died following transport was recorded as condition 2, dead on day 1.

Retained fish were held in 3000-4000L tanks at both MRL and a commercial fisher's holding tanks (Nubeena) while a larger tank of approximately 15 000L was partitioned into approximately 3000L sections for use at the fish processing facility (St. Helens). The MRL tanks were subject to flow-through seawater while recirculating water systems were used in the other locations. Air stones were used in all locations to ensure adequate oxygenation was achieved. Fish were held for seven days. A mixture of mussels, abalone viscera, live crabs and commercial salmonid pellets were used to feed fish during the holding period. The influence of temperature, condition, net soak time, fishing depth on survival were investigated using GLM for each species. Potential confounding factors such as handling methods between the two fishers and researcher, transportation times from different locations and holding tanks were not explicitly investigated but were assumed to be included in an additional factor termed 'fisher'. Differences between 'fisher' (called St. Helens, MRL and Nubeena) were assessed, and, if significant, would indicate confounding due to one or more of these factors.

Individual species' survival rates (by condition) were multiplied by the proportion of discards (per species) in commercial catches (Chapter 2) to estimate total discard survival. Total discard mortality, the converse of survival, was then summed with the proportion of individuals that were retained for sale to calculate the total fishing-induced mortality.

4.3 RESULTS

4.3.1 Fish condition

With the exception of bastard trumpeter and blue throat wrasse, the majority of individuals for each species were in condition 1 following capture, with no visible external damage. Blue throat wrasse comprised the highest proportion of fish in poor condition (condition 3, Table 4.2), largely due to major scale loss, bleeding and spine damage. Individuals in condition 4 (dead at capture) were rarely encountered for all species.

Table 4.2 Condition frequency by proportion and sample size of banded morwong (*Cheilodactylus spectabilis*, BMW), purple wrasse (*Notolabrus fucicola*, PUW), bastard trumpeter (*Latridopsis forsteri*, BTR), marblefish (*Aplodactylus arctidens*, MBL), draughtboard shark (*Cephalocyllium laticeps*, DS), long-snouted boarfish (*Pentaceropsis recurvirostris*, LSB) and blue throat wrasse (*N. tetricus*, BTW). All fish were captured by commercial banded morwong fishing vessels on the east coast of Tasmania between 2005 and 2006.

Condition	BMW	PUW	BTR	MBL	DS	LSB	BTW
1	0.90	0.62	0.35	0.96	0.98	0.70	0.44
2	0.09	0.21	0.55	0.03	0.01	0.30	0.27
3	0.01	0.15	0.10	0.01	<0.01	0	0.27
4	0	0.02	0	0	0	0	0.02
Sample size	2212	66	199	956	617	69	62

Fish condition was assessed with regard to fishing depth, soak time, fish length, season and species using GLM. The analysis revealed that species ($F = 6.759$, $df = 6$, $p < 0.000$) and season ($F = 3.868$, $df = 3$, $p = 0.009$) were significant factors with a residual df of 825, all other factors were non-significant. Further investigation into differences in condition between species showed all pairwise comparisons were significant ($p< 0.05$), except bastard trumpeter and blue throat wrasse ($p= 0.828$), and marblefish and draughtboard shark ($p=1.000$). Similarities between bastard trumpeter and blue throat wrasse were due to relatively high proportions of individuals with major bruising and scale loss (condition 2 and 3). In contrast, marblefish and draughtboard shark were similar due to the high proportion of individuals displaying no obvious physical damage following capture (condition 1).

Investigation into seasonal differences in fish condition following capture highlighted a significant interaction between season and species ($F = 5.890$, $df = 18$, residual df = 825, $p < 0.001$) due to bastard trumpeter alone. A pairwise comparison between seasons for bastard trumpeter revealed a significant difference between summer and all other seasons ($p < 0.0001$), with overall better condition during summer ($F = 7.554$, $df = 3$, residual $df = 28$, $p = 0.01$).

4.3.2 Aquaria trials

Using GLM, condition was found to have a significant effect on the survival rate of banded morwong ($F = 3.184$, $df = 2$, residual $df = 89$, $p = 0.046$), purple wrasse ($F = 10.838$, $df = 2$, residual $df = 17$, $p = 0.001$) and blue throat wrasse ($F = 6.741$, $df = 2$, residual $df = 20$, $p = 0.006$). In addition, the survival rates of bastard trumpeter and long-snouted boarfish decreased between conditions 1 and 2, and conditions 2 and 3 (Table 4.3); however, these reductions in survival were found to be non-significant. The analysis was not undertaken for marblefish and draughtboard shark as all individuals from these species survived for the duration of the experiment.

'Fisher' was a significant factor in the survival of banded morwong ($F= 48.600$, $df= 2$, residual $df = 18$, $p< 0.000$), bastard trumpeter ($F= 39.288$, $df= 2$, residual $df = 18$, $p< 0.000$) and blue throat wrasse ($F= 8.116$, $df= 2$, residual $df = 18$, $p= 0.003$); however, was not significant for any other species. The significant difference between 'fisher' for banded morwong survival was due to two banded morwong deaths from 72 individuals (3%) at St. Helens. In contrast, no mortalities were observed in banded morwong at MRL or Nubeena. The survival of bastard trumpeter in St. Helens was found to be significantly different to MRL and Nubeena. In addition, a significant difference was found between 'fisher' in all locations for the survival of blue throat wrasse. No other significant factors, including temperature, were found for any species.

Discard mortality rates were highest in long-snouted boarfish; however, due to their high rate of release, this species had the lowest fishery-induced mortality of the commercially important species (Table 4.3). In contrast, blue throat wrasse had a relatively high rate of discard mortality as well as the highest fishery-induced mortality. The fishery-induced mortality for banded morwong was relatively low, despite being the target of the commercial fishery. This occurred as the majority of captured individuals were in good condition and had a low discard mortality rate.

Table 4.3 Aquaria trial statistics. The proportions (by number) of commercially retained and discarded species were obtained from onboard commercial catch sampling and assume adherence to legal size limits. All fish were retained for a period of seven days. Retained individuals were included as mortalities in the calculation of fishery-induced mortality.

Species	Sample size	Condition grade	Survival rate by condition	Discarded fish by condition (% total catch)	Total mortality of discards (%)	Fishery-induced mortality (%)
Banded	77	1	1.00	42.13		
morwong	13	2	0.92	4.16		
(*Cheilodactylus*	2	3	0.50	0.70	4.17	54.96
spectabilis)						
Purple wrasse	10	1	1.00	29.82		
(*Notolabrus*	8	2	0.25	10.18		
fucicola)	2	3	0.50	7.27	6.18	54.97
Bastard	15	1	0.33	6.24		
trumpeter	13	2	0.31	9.86		
(*Latridopsis*	2	3	0.00	1.99	7.11	83.28
forsteri)						
Marblefish	18	1	1.00	95.50		
(*Aplodactylus*	6	2	1.00	3.14		
arctidens)	0	3	-	1.36	0.00	0.00
Draughtboard	25	1	1.00	98.38		
shark	0	2	-	1.46		
(*Cephaloscyllium*	0	3	-	0.49	0.00	0.00
laticeps)						
Long-snouted	4	1	0.50	55.65		
boarfish	6	2	0.27	25.35		
(*Pentaceropsis*	0	3	-		34.67	47.74
recurvirostris)						
Blue throat	7	1	0.86	6.97		
wrasse (*N.*	9	2	0.78	4.39		
tetricus)	7	3	0.00	4.39	13.81	86.21

4.4 DISCUSSION

Capture condition was found to significantly effect the survival of banded morwong, blue throat wrasse and purple wrasse. This is of particular importance as individuals from each of these species are retained for live sale in the fishery. The use of condition as a proxy for survival (Patterson et al. 2000, Rudershausen et al. 2007) may therefore be used to estimate the rate of discard survival and fishery-induced mortality (Harris 1995) for these species. Information on condition at capture could be beneficial in the banded morwong fishery as the majority of banded morwong and wrasse are sold live and profits are dependent on the survival of these retained fish prior to sale. An understanding of the potential survival rate of individuals would enable fishers to calculate the efficacy of retaining fish in condition grades 2 and 3 for live sale. This information would then provide an early indication of which individuals could be profitably marketed live.

The use of control fish may have been beneficial to reduce the number of confounding factors in the aquaria trials (Wilde et al. 2003). Logistical constraints restricted our ability to capture control fish in addition to treatment fish. Controlling for the effect of transport and aquaria may have created further problems because confining fish to cauffs (sea-cages) would have been necessary to avoid transport. Confinement in cauffs may then have exposed the fish to additional confounding factors, such as stress through seal harassment and swell, which have been shown to negatively impact the survival of cauff-retained fish (Rimmer and Franklin 1997, Davis 2005). As it was impossible to avoid additional confounding factors, cauffs were not used in this study. The effect of factors, such as handling and different tanks on survival rate, while incorporated into the factor 'fisher', could not be established. Yet, the lack of a significant effect for 'fisher' on the majority of species indicated that these potentially confounding factors may have had a negligible impact on survival or, more probably, may have had an equally detrimental impact on species survival for all fishers.

The target species, banded morwong, were found to have relatively high survival rates and consistency with survival rates from a commercial processing facility would provide further evidence that these results are representative of commercial fishing practices. Such consistency is important in order to partially address some of the confounding factors through handling and transport that were associated with this study. Banded morwong retained at commercial processing facilities may be assumed to be in condition 1 and 2. This is because fish in poorer conditions tend to be selectively removed by fishers prior to transportation. Further selection based on condition then occurs on arrival at processing facilities such that the vast majority of individuals will be in healthy (condition 1), with a very small proportion of condition 2 fish included (S.J. Metcalf, unpublished data). The survival rate for banded morwong held by processors could be expected to fall between the observed condition 1 and 2 survival rates from this study, i.e. 100% survival for condition 1 and 92% survival for condition 2. In fact, survival rates for commercially caught banded morwong held for at least a week in a major commercial fish processing facility, were generally consistent with the findings of this study with an overall survival rate of 96% (n = 3181) (S.J. Metcalf, unpublished data). While there were a number of potentially confounding factors in this study, these results suggest that the survival rates for banded morwong may be a reasonable representation of the survival rate experienced in the commercial fishery for both retained (for live sale) and discarded fish (based on legal size limit requirements).

The relatively high survival rate in banded morwong is not surprising given that gear type, capture and transport methods have been specifically developed in live fish fisheries to ensure high survival rates in the target species. Thus, while the use of large mesh gillnets to capture banded morwong for live sale may be in the best interests of maintaining the target species in good condition, they may have differential effects on non-target species. For instance, marblefish, draughtboard shark and banded morwong had a high rate of survival while purple wrasse and long-snouted boarfish had a moderate rate of survival. Mortality of non-target species has been found to be species-specific by gear type in some live fish fisheries (Chopin and Arimoto 1995) as well as other fisheries (Serafy et al. 2009). Gillnet caught wrasse, in particular blue throat wrasse, experience relatively poor survival rates. As a consequence, the commercial live fish wrasse fishery utilises fish traps, and hook and line (Ziegler et al. 2008), as these methods have less of an impact on wrasse survival than was observed for this species in the banded morwong fishery.

Fisheries-induced mortality may have consequences for individual species through recruitment and reproduction. However, uncertainty still remains as to the precise impact of the banded morwong fishery on the ecosystem. In some situations discard mortalities, in addition to the retained catch, have been proposed as responsible for altering size distributions within fish populations, as well as species composition and diversity (Pauly et al. 2001, Christensen et al. 2003). Yet, in the banded morwong fishery the catch rates of non-target species with low survival rates were also low. One method to address the uncertainty of ecosystem effects is to use ecosystem models to investigate multi-species interactions (Coll et al. 2008). Yet, the production of ecosystem models that involve fishery impacts may be hampered if specific information on the catch and mortality rates of non-target species is unavailable. Estimates of the overall fishing mortality of target and non-target species through direct observation or the use of condition as a proxy, as in this study, can therefore be useful to provide information on overall fishery-induced mortality for ecosystem modelling. In addition, the use of condition indices may be of particular value in the live fish trade as they allow fishers to select individuals with reduced probability of survival to market yet could maintain some economic value if sold fresh, thereby reducing wastage.

These investigations are important as the survival of retained and discarded target species is integral to the sustainability of banded morwong fishery profits. The estimates of discard mortality from this study will be used in ecosystem models of the inshore reefs of eastern Tasmania to investigate the effects of the banded morwong fishery on both target and non-target species as well as other elements of the ecosystem (Chapter 6).

5. IMPORTANCE OF TROPHIC INFORMATION, SIMPLIFICATION AND AGGREGATION ERROR IN ECOSYSTEM MODELS

5.1 INTRODUCTION

Ecosystem models may be quantitative or qualitative and can aid conventional stock assessment by providing information on the state of the ecosystem. Trophic linkages often form the backbone of ecosystem models and are commonly inferred from dietary analysis (Christensen and Pauly 1992, Okey et al. 2004a). The calculation of traditional metrics, such as dietary overlap and niche breadth using data collected from the focal system, can highlight important ecological factors including trophic guilds and ontogenetic variation (Munoz and Ojeda 1998). This information can then be incorporated into ecosystem models. Different trophic web structure can result in different model predictions and may have a large impact on research and management decisions (Pinnegar et al. 2005). This was demonstrated by Yodzis (2001) when studying the effect of seal culling on fisheries. One model suggested the fishery would benefit from a cull and another, which included different trophic linkages, did not support this conclusion. In order to produce model predictions that will be useful in resource management, the trophic web most closely aligned to the natural system should be used. Without this information, management strategies may be based on predicted responses that do not occur in the natural system or may not predict responses that do occur.

The inclusion of the entire food web into models may provide a more realistic view of the system than aggregated models; however, large complex models often contain too much uncertainty (Raick et al. 2006) for their use to be beneficial. Species aggregation is often necessary to provide useful predictions (Auger et al. 2000, Fulton et al. 2003, Raick et al. 2006). Many alternative methods of aggregating ecosystem variables may be used, for instance, in many studies using Ecopath (e.g. Christensen et al. 2003, Okey et al. 2004a, Bulman et al. 2006) functional groups were aggregated according to the impact of commercial fisheries and life history characteristics, such as size. In addition, variable aggregation may be based on turnover rates and stock size (Gardner et al. 1982) and, commonly, on the modeller's perception of the system (Luczkovich et al. 2002). In other studies, more mathematical approaches such as Bray Curtis similarities (Bray and Curtis 1957, Pinnegar et al. 2005), Euclidean distance (Sokal and Sneath 1963) and regular equivalence (Luczkovich et al. 2003), have been used to aggregate large numbers of species into groups. In contrast to complex models, species aggregation for simplification may result in the model becoming oversimplified (Raick et al. 2006). In addition, following the aggregation of species in any model, parameter uncertainty may still remain due to the dynamic nature of the real world. A compromise between high uncertainty and oversimplification may be necessary and can be achieved by reducing the rate of error between detailed and simplified model predictions.

Much discussion has occurred on the type and value of aggregation techniques (e.g. Raffaelli and Hall 1992, Raick et al. 2006). This discussion has focussed on methods of aggregation, such as "perfect aggregation", in which the dynamics of the aggregated variable are consistent with the dynamics of the disaggregated variables (Iwasa et al. 1987). This method was later found to be too restrictive for many forms of ecosystem analyses and the focus shifted instead to the "best approximate aggregation" where the minimum inconsistency of dynamics between aggregated and disaggregated variables was preferred (Iwasa 1989). Minimum inconsistency was desired as the aggregation of variables may result in prediction error due to the loss of information from disaggregated to aggregated groups (Auger et al. 2000). For example, information regarding the specific prey items of each species in an aggregated variable may be lost. Error may also occur due to the aggregation of predators and their prey (Gardner et al. 1982, Fulton et al. 2003), and of variables with differing turnover times (O'Neill and Rust 1979, Gardner et al. 1982). The importance of uncertainty due to aggregation error has been acknowledged (Gardner et al. 1982, Cale et al. 1983) yet is rarely discussed in modelling studies with aggregated variables. This is not a trivial problem, particularly if the models are then used to inform management decisions. In addition, alternative methods of aggregation may result in different levels of error. If we assume the disaggregated model is a reasonable representation of the ecosystem, the use of the aggregation method that creates the least error can increase the predictability of the model results.

Qualitative ecosystem models, using signed digraphs (Levins 1974, Levins 1975), may be used to generate predictions and guide further ecosystem studies. These models can also provide a means to analyse aggregation error and are a quick and efficient way to study ecosystem structure. Aggregation error must be investigated following the simplification of models through variable aggregation to ensure the model predictability is retained. As the use of ecological models in management requires the assumption that the model is a reasonable representation of the system, the production of simplified models with high certainty of prediction of disaggregated model results is important. This is also necessary to ensure the predictions generated can be of use to management.

Qualitative modelling was used to produce a detailed initial model on the basis of the ecosystem-specific dietary information obtained during this study. Dietary information was collected for banded morwong, blue throat wrasse (*Notolabrus tetricus*), purple wrasse (*N. fucicola*), bastard trumpeter (*Latridopsis forsteri*), long-snouted boarfish (*Pentaceropsis recurvirostris*) and marblefish (*Aplodactylus arctidens*). This information was required because there were very few published accounts of the diets of these commercially important reef fish. Furthermore, available studies do not include information on commonly caught size classes (Choat and Clements 1992) or have low sample sizes for the species of interest (Fenton 1996, Bulman et al. 2001). The variables in this model were aggregated using Euclidean distance, Bray Curtis similarity and regular equivalence, and included in three separate simplified models. Comparisons between qualitative models with and without variable aggregation can be used to assess the level of aggregation error produced by different methods. This study explored how dietary information may be utilised to construct ecosystem models using different methods of model simplification. While the models and variables aggregated in this study were based on trophic linkages, the methods of aggregation and investigation into aggregation error may be used for other types of data, such as turnover rates and production rates. This study also illustrates the power of qualitative analysis as a method of ecosystem investigation for small-scale fisheries.

5.2 METHODS

5.2.1 Fish collection and processing

Each of the six study species is commonly captured in the Tasmanian live fish fishery for banded morwong, which uses large mesh nets of 115 and 140 mm stretched mesh. During onboard commercial catch sampling, banded morwong were found to constitute approximately 35% of the total numbers of fish caught (Chapter 2). Blue throat and purple wrasse, bastard trumpeter and long-snouted boarfish are byproduct species and constitute a further 19% of the catch. Marblefish is a bycatch species that comprises around 21% of the catch. Byproduct species are retained non-target species and in this fishery the commercial catch is restricted by species-specific size limits. In contrast, bycatch are species without commercial value and are not retained by fishermen.

Fish were collected for dietary analysis using gillnets with stretched mesh sizes ranging between 64 and 140 mm to ensure that adequate sample sizes of each species were captured. Collections were undertaken at various sites within the same region as the commercial banded morwong fishery on the east coast of Tasmania (Fig. 5.1). Banded morwong were collected between 2004 and 2006, while all other species were collected in 2005-2006. A general understanding of the overall diet is sufficient for many ecosystem models and particularly when using broad qualitative models. As a result, diet was not assessed by individual location or season. Each fish was measured (fork length), weighed and sexed prior to gut removal.

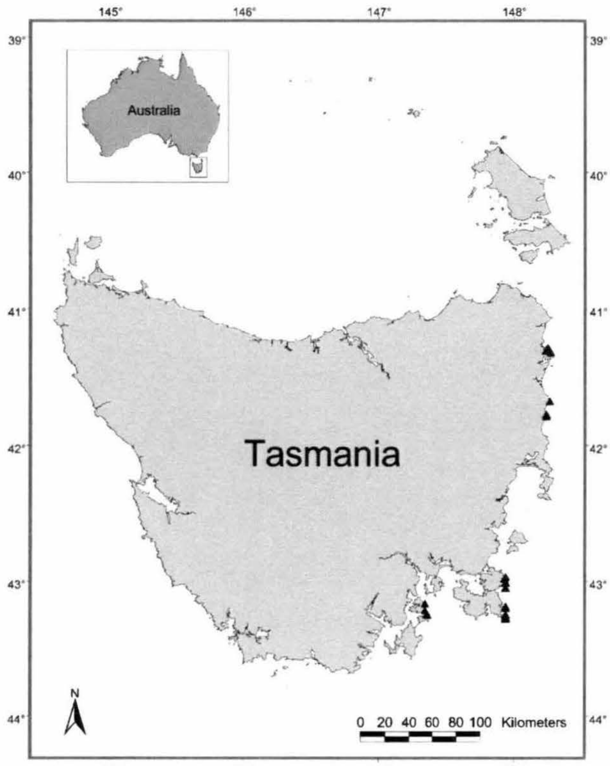


Figure 5.1 Fish collection locations on the east coast of Tasmania (n = 23 locations) during 2004-2006.

The contents of the entire gut (stomach and intestines) were collected for all species to ensure that no bias occurred between species with and without defined stomachs (marblefish). Each sample was stored in 70% ethanol, dried with paper towel and weighed to the nearest 0.01 g prior to examination. The majority of fish examined (> 90% per species) had food in the gut and only these fish have been included in the dietary analyses. Gut contents were passed through a 1 mm sieve and identified, where possible, to species level under a dissecting microscope. Individual prey groups were weighed separately to the nearest 0.01 g for each stomach.

The number of samples necessary to obtain an accurate estimate of species richness within the diet was investigated using sample-based rarefaction in EstimateS (Version 7.5.0). Species accumulation curves were calculated by re-sampling observed data. This established that the sample sizes utilised for each species were higher than the minimum required to closely approach the asymptote of the curve describing the number of prey items versus the number of samples (Colwell et al. 2004).

Diet indices

The percent number (% N), percent weight (% W) and percent frequency of occurrence (% FOO) were calculated in addition to the percent index of relative importance (% IRI) (Hyslop 1980). The % IRI was calculated as it may reduce potential biases caused by the use of a single metric. Prey items were grouped into higher taxa groupings (Order, Phylum) to calculate the diet metrics. This was necessary to compare the diet composition between fish species and to reduce bias between individuals with easily identifiable prey items and those in which prey were rarely identified (Cortes 1997).

5.2.3 Dietary similarity and overlap

Diet similarity and overlap were calculated between species. In order to determine the level of dietary similarity, multidimensional scaling (MDS) was undertaken using the prey type and number of prey items consumed. The data were square root transformed to reduce the effect of the dominance prey type and Bray-Curtis dissimilarities were used for MDS in Primer 5 (Version 5.2.9).

The simplified Morisita's index (*sensu* Horn 1966) using % W was calculated to determine the level of overlap between species. Significant dietary overlap was assumed when Morisita's index was greater than 0.6 (Wallace and Ramsey 1983). Percent weight was used to determine which prey species should be included in the model as qualitative modelling and many types of ecosystem modelling are mass-based. Higher taxonomic groupings (Order and Family) were used for all overlap analyses.

5.2.4 Ecosystem models based on dietary information

Qualitative analysis was used to investigate ecosystem structure and aggregation error produced through different aggregation methods. A detailed initial model was constructed on the basis of dietary information obtained during the present study and from a review of the literature (Table 5.1). To ensure that the models did not become unwieldy with too many variables, only prey that constituted greater than 15% W per fish species were included in each model. This level of detail was selected because it includes the prey groups that constitute the majority of the diet for each species.

Table 5.1 References used for dietary information included in the initial model. A Tasmanian study was used in preference to other available information. Unreported sample sizes are referred to as NA.

Species	Location	Sample size (n)	Reference
Seals	Tasmania	977	Hume et al. 2004
Gastropods	Tasmania	NA	Edgar 2000
Decapods	New Zealand	724	Woods 1993
Amphipods	Tasmania	NA	Edgar 2000
Ophiuroids	Tasmania	NA	Edgar 2000
Bivalves	Tasmania	NA	Edgar 2000
Isopods	Tasmania	NA	Edgar 2000
Polychaetes	South Australia and Victoria	NA	Holloway and Keough 2002
Sponges	Tasmania	NA	Edgar 2000
Bryozoans	Western Australia	NA	Lisbjerg and Petersen 2000

The simplification of the initial model was undertaken using the qualitative diet matrix to create three different models using Euclidean distance (ED), Bray Curtis (BC) similarity and regular equivalence (RE) through the REGE algorithm (Luczkovich et al. 2003, www.analytictech.com/downloaduc6.htm). Similarity matrices were produced using these measures and used to determine which variables to aggregate. Euclidean distance (ED) is the shortest distance between variables in ecological space (Sokal and Sneath 1963). This distance measure does not take abundance into account and treats every number equally. Euclidean distance measures presence and absence unlike Bray Curtis (BC) similarity, which also measures magnitude through relative abundances in the data (Bray and Curtis 1957). Neither euclidean distance nor Bray Curtis similarity can use the negative values produced as a result of the mortality of species through predation. In contrast, regular equivalence (RE) takes both negative values from predators and positive values from prey into account in the calculation of similarity (Luczkovich et al. 2003). This method is not, however, sensitive to relative abundances in the data. The REGE algorithm uses species-by-species matrices to calculate species-by-species matrices of **R** coefficients, which measure the regular equivalence between variables (Luczkovich et al. 2003). These coefficients have ordinal properties that may then be visualised using multivariate statistics.

A qualitative diet matrix was used in the calculation of Bray Curtis similarities, Euclidean distance and regular equivalence, where the prey of a variable was denoted by a positive sign (+1) and predators of the variable were denoted by a negative sign (-1). This matrix was used instead of actual dietary metrics such as % W because we were interested in the qualitative presence/absence of linkages between species. Dendrograms were produced to display the similarity and distance measures for all three methods to visualise variable aggregations. The number of clusters selected for use in a model can be based on a desired level of equivalence or similarity among members of a cluster. The desired level of equivalence may vary given the questions being asked of the models and when comparing between models, as in this study, the level of aggregation should be similar. If this does not occur, comparisons will be meaningless. In this study, models of ten or eleven variables were selected for use, as this level of aggregation remained biologically meaningful. In contrast, models of this system containing only 4 or 5 variables would be inappropriate, as they would require the aggregation of predators and prey, as well as placing detritus with invertebrates.

Following the simplification of a model through the aggregation of variables, the number of predictions that were consistent between the simplified (aggregated) and initial (not aggregated) model were compared and used to assess aggregation error. We illustrate this with an example of a small food web (Fig. 5.2a). In this example, only two methods, Euclidean distance (ED) and regular equivalence (RE) were used to aggregate the models. Aggregation using Bray Curtis similarities follows the same methods and was therefore not included in the example. In this model, one species (3) is both a predator and a prey while another two species (2 and 4) at the same trophic level have no predators in the system. Species interactions are described in the community matrix (**A**) (Table 5.2). The matrix of predictions (adj (-**A**)) is in Table 5.3 where, for example, an increase to species 2 will have a negative effect on the abundance of species 1. The asterisks in Table 5.3 denote variables, which produce incorrect predictions when aggregated using regular equivalence in Table 5.4.

Table 5.2 Species interactions in the community matrix (**A**) for Figure 5.2a.

Predictions/ Perturbations	*Species 1*	*Species 2*	*Species 3*	*Species 4*	*Species 5*
Species 1	-	0	+	0	0
Species 2	0	-	0	0	+
Species 3	-	0	-	0	+
Species 4	0	0	0	-	+
Species 5	0	-	-	-	-

Table 5.3 Adjoint of the negative community matrix (-**A**) for Figure 5.2a showing the predictions of response (across rows) to perturbation (down columns). Asterisks denote variables that produce incorrect predictions when aggregated using regular equivalence (Table 5.4).

Predictions/ Perturbations	*Species 1*	*Species 2*	*Species 3*	*Species 4*	*Species 5*
Species 1	+	- *	+	- *	+
Species 2	+	+	-	-	+
Species 3	-	-	+	-	+
Species 4	+	-	-	+	+
Species 5	+	-	-	-	+

Regular equivalence (RE) (Luczkovich et al. 2003) has been used in social network theory and takes both predator and prey links into account. Using this method, species 3 remained separate from species 2 and 4 (Fig. 5.2b). This resulted in the following matrix of predictions (Table 5.5).

Table 5.4 Adjoint of the negative community matrix (-**A**) for Figure 5.2c with species aggregated using regular equivalence. The predictions of response (across rows) to perturbation (down columns) are shown. '^' highlights the positive response species 1 has following a perturbation to the aggregated variable *Sp. 2,4*.

Predictions/ Perturbations	*Species 1*	*Species 2,4*	*Species 3*	*Species 5*
Species 1	+	+ ^	+	-
Species 2,4	-	+	+	-
Species 3	+	-	+	-
Species 5	+	-	+	+

In contrast, using Euclidean distance (ED) to aggregate model variables, all middle trophic level species (2, 3 and 4) would be aggregated together (Fig. 5.2c) and result in the matrix of predictions seen in Table 5.5. As a result, the difference between species 3 and species 2 and 4 is not evident using this method. Dashed lines surround the aggregated species.

Table 5.5 Adjoint of the negative community matrix (-**A**) for Figure 5.2b with species aggregated using regular equivalence. The predictions of response (across rows) to perturbation (down columns) are shown. '^' highlights the positive response species 1 has following a perturbation to the aggregated variable *Sp. 2,4*.

Predictions/ Perturbations	*Species 1*	*Species 2,3,4*	*Species 5*
Species 1	+	+	+
Species 2,3,4	-	+	+
Species 5	+	-	+

To calculate the number of predictions in the simplified models that were consistent with the detailed model, the signs (+, -, 0) of the variables to be aggregated from the detailed model need to be summed. The sign of the sum is then the correct prediction for the aggregated group. For instance, in Table 5.2, increases to species 2 and 4 (columns) create negative responses in species 1(row) (indicated by *). As a result, the sum of these signs (- plus -) is also negative. In the matrix of predictions for the simplified model (Table 5.4), species 2 and 4 (column) produced a positive effect on species 1 (row) (indicated by ^). The prediction in the simplified model is therefore inconsistent with the detailed model and will contribute to the aggregation error.

In this study, the percent of predictions consistent with the detailed model were compared between the RE, BC and ED models. The percentage of signs that were not consistent between the simplified and detailed model were reported as aggregation error.

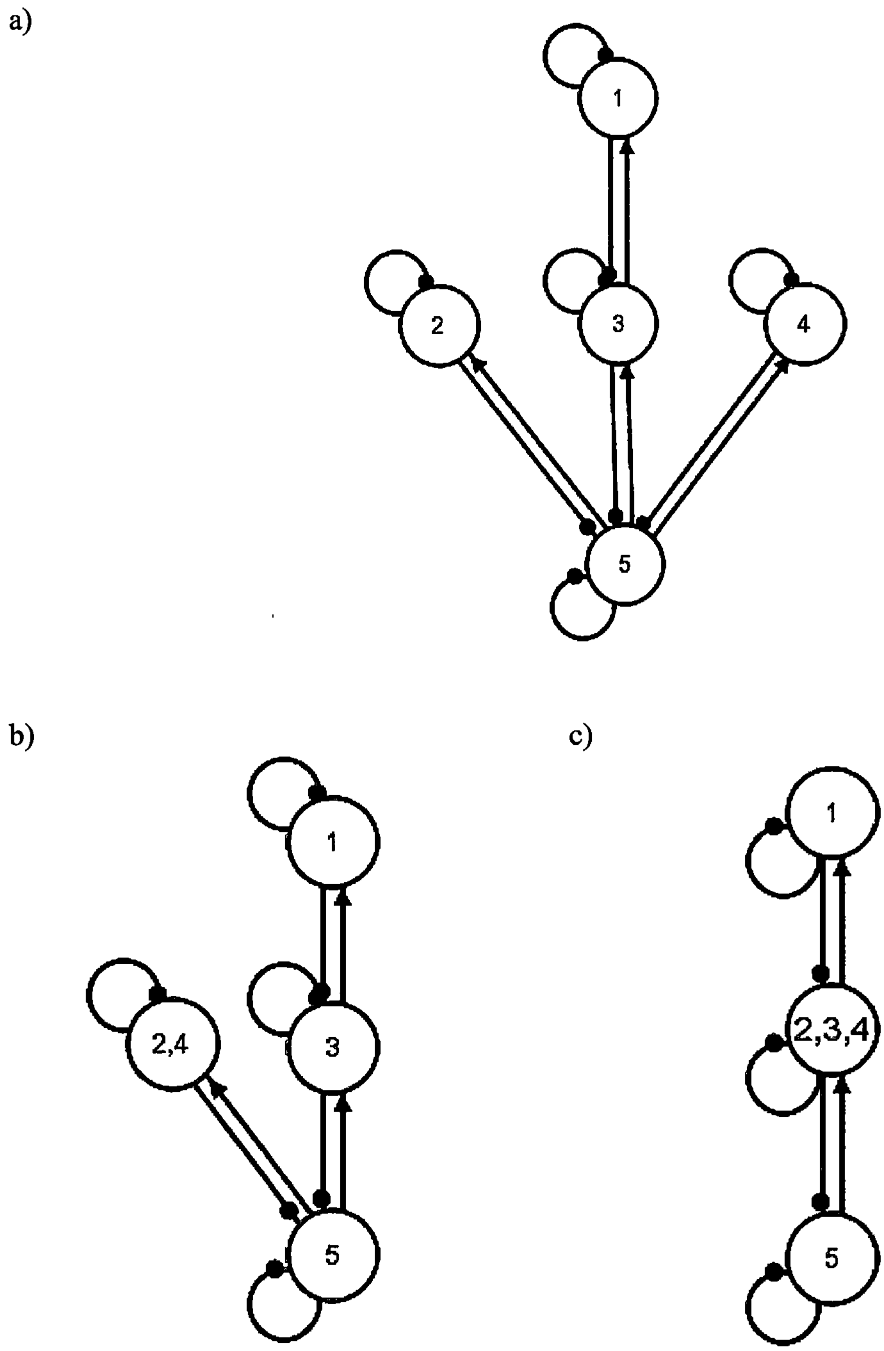


Fig. 5.2 a) Example of a detailed food web model where closed circles represent negative effects and arrows represent positive effects, b) model aggregated using regular equivalence which resulted in the aggregation of variables 2 and 4 alone and c) model aggregated using Euclidean distance which resulted in the aggregation of variables 2,3 and 4.

5.3 RESULTS

The diets of 227 individuals from the six study species were examined with a total of 74 prey items identified. These prey items were placed into higher taxonomic groups (Order, n= 17) for inclusion in models. Minimum sample sizes for species richness were calculated using sample-based rarefaction and ranged between 10 (banded morwong) and 30 (long-snouted boarfish) individuals per species. Actual sample sizes were: banded morwong (n = 62); bastard trumpeter (n = 44), blue throat wrasse (n = 30); purple wrasse (n = 24); long-snouted boarfish (n = 41); and marblefish (n = 26).

5.3.1 Dietary analyses

Prey items that comprised greater than 15% W have been reported as these prey species were included in the models. Detail on alternative metrics and specific prey items of the study species are shown in Table 5.2.

5.3.2 Dietary overlap

MDS clearly separated the diets of marblefish and long-snouted boarfish diets from the remaining diets of the remaining fish species, grouped here as benthic invertebrate feeders (Fig. 5.3). The similarity between benthic invertebrate feeders was obvious with clearly overlapping ellipses. Banded morwong and bastard trumpeter displayed a significant overlap due to the reliance on decapods and amphipods. This was supported through the use of Morisita's Index at 0.69. The dietary overlap between blue throat wrasse and purple wrasse was also significant (0.68) due to a reliance on bivalves, particularly *Mytilus edulis*.

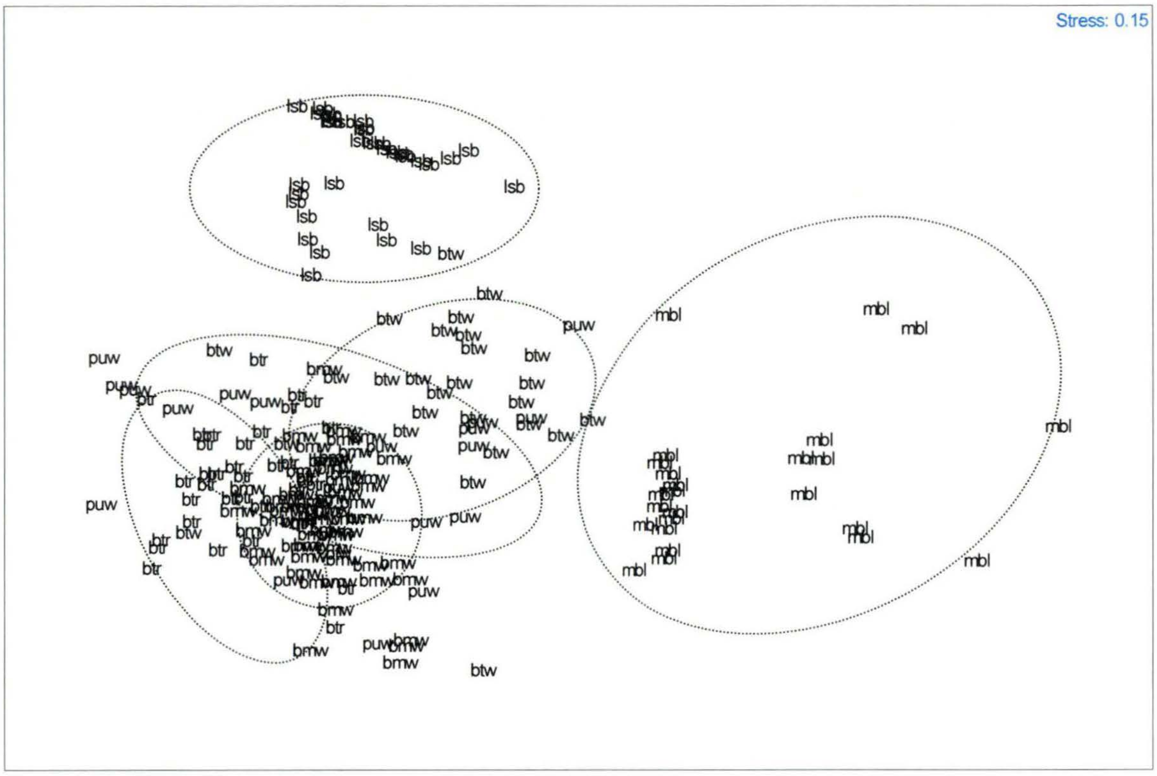


Figure 5.3 MDS plot of prey item similarity for fish samples collected on the east coast of Tasmania. Each ellipse encloses almost all the specimens of one species and highlights the separation between species: mbl- marblefish (*Aplodactylus arctidens*); lsb- long-snouted boarfish (*Pentaceropsis recurvirostris*); btr- bastard trumpeter (*Latridopsis forsteri*); puw- purple wrasse (*Notolabrus tetricus*); btw- blue throat wrasse (*N. tetricus*); bmw- banded morwong (*Cheilodactylus spectabilis*). Sample sizes are provided in the text.

Table 5.2 Prey diet metrics: % N- percent number; % FOO- percent frequency of occurrence; and % W- percent weight. Higher order groupings used for diet overlap calculations are in bold. Species are: BMW, banded morwong (*Cheilodactylus spectabilis*)(n = 62); BTR, bastard trumpeter (*Latridopsis forsteri*)(n = 44); BTW, blue throat wrasse (*Notolabrus tetricus*)(n = 30); PUW, purple wrasse (*Notolabrus fucicola*)(n = 24); LSB, long-snouted boarfish (*Pentaceropsis recurvirostris*)(n = 41); and MBL, marblefish (*Aplodactylus arctidens*)(n = 26).

Prey Items	BMW				BTR				BTW				PUW				LSB				MBL			
	% N	% FOO	% W	% IRI	% N	% FOO	% W	% IRI	% N	% FOO	% W	% IRI	% N	% FOO	% W	% IRI	% N	% FOO	% W	% IRI	% N	% FOO	% W	% IRI
GASTROPODA	**2.6**	**62.9**	**0.1**	**1.5**	**8.8**	**86.0**	**3.4**	**6.0**	**27.5**	**90**	**0.9**	**38.9**	**50.1**	**87.5**	**27.0**	**52.4**	**4.2**	**7.3**	**0**	**0.2**	**0**	**3.8**	**0**	**0**
Retusidae	0 3	23 3	0	0																				
Maoricolpus roseus	0	1 7	0	0																				
Haliotis spp	0 1	8 3	0 1	0	0	2 3	0 2	0					0.1	4.2	0 8	0 1								
Limpet	0 1	8 3	0	0					0 5	6 7	0 9	0 1	6 1	25 0	4 6	4 3								
Trochidae									0 4	3 3	0	0												
Gastropods spp	2 1	55 0	4 5	36 6	8 7	86 0	3 1	5 9	26 6	86 7	0	38 8	8 9	70 8	22 5	35 9	4 2	7 3	0	0 2	0	3 8	0	0
AMPHIPODA	**55.9**	**95.2**	**30.8**	**47.9**	**35.7**	**100**	**23.8**	**30.5**	**20.9**	**36.7**	**0**	**12.0**	**22.4**	**45.8**	**3.8**	**8.3**	**9.1**	**9.8**	**0**	**0.5**	**0**	**0**	**0**	**0**
Podoceridae	1 9	23 3	0 8	0 4	0 3	18 6	0 1	0					1 2	16.7	0	0 3	0 7	2 4	0	0				
Amphilochidae					0 2	2 3	0 2	0																
Gammaridae/Caprellidae/ Melitidae	53 9	95 0	30	52 7	35 2	100	23 6	29 4					19 2	45 8	3 8	7 9	8 4	7 3	0 1	0 4				
Paramelitidae	0 1	5 0	0	0																				
Unknown amphipod									20 9	36 7	0	11 0												
DECAPODA	**13.9**	**88.7**	**32.1**	**24.5**	**20.9**	**100**	**46.4**	**36.0**	**24.9**	**90**	**9.5**	**35.9**	**10.9**	**70.8**	**5.3**	**8.9**	**8.6**	**26.8**	**0.1**	**1.3**	**0**	**0**	**0**	**0**
Alphaeidae	4 8	70	5 9	5 0	5 1	74.4	4 7	4 2	1 4	16 7	0 1	0 3	1 9	20 8	0 2	0 7								
Halicarcinus ovatus					4 2	60 5	8 1	4 3	12 8	30	0	0					1 3	7.3	0 2	0 1				
Elamena abrolhensis	6 6	86 7	21 4	16 2																				
Naxia spinosa	0 2	18 3	2 4	0 3																				
Notomithrax ursus					1 1	48 8	2 4	1 0																
Galathea australiensis	1 8	20	0 4	0 3	0 1	7 0	0 1	0																
Unknown brachyurid	0 1	5 0	0	0																				
Unknown decapod	0 4	26 7	1 8	0 4	9 5	93 0	27 2	19 8	10 3	80	2 9	17 8	9 0	66 7	0	9 7	3 0	4 9	0	0 1				
Paguridae	0	3 3	0 2	0	1 0	25 6	1 1	0 3	0 4	3 3	6 5	0 3					3 6	12 2	0	0 3				
Palaemonidae																	0 7	2 4	0	0				
BIVALVIA	**16.7**	**90.3**	**9.0**	**10.2**	**14.2**	**83.7**	**2.4**	**7.4**	**17.2**	**15**	**31.5**	**12.9**	**13.5**	**50.0**	**51.7**	**25.3**	**3.3**	**7.3**	**0**	**0.1**	**0**	**3.8**	**0**	**0**
Glycymeris striatularis	0 2	5 0	0	1 0																				
Mytilus edulis									10 2	26 7	30 2	11 6	12 1	29 2	4 5	7 8								
Unknown bivalve	13 0	88 3	5 8	11 1	13 8	81 4	2 2	7 6	7	63 3	1 3	9 3	1 4	33 3	0	0 8	3 3	7 3	0	0 1	0	3 8	7 2	0 4
Mesoplepum tasmanicum	0 3	18 3	0 2	0 1																				
Panopea australis	3 2	61 7	3 2	2 7																				
OPHIUROIDEA	**0.2**	**17.7**	**2.8**	**0.3**	**1.0**	**27.9**	**1.8**	**0.5**	**0.6**	**6.7**	**8.7**	**0.1**	**0.1**	**4.2**	**0.1**	**0**	**68.4**	**100**	**99.6**	**97.3**	**0**	**3.8**	**0**	**0**
Ophiactis resiliens	0 2	17 7	2 8	0 3	1 0	27 9	1 8	0 5									68 4	100	99 6	97 3	0	3 8	0	0
ECHINOIDEA	**0.2**	**19.4**	**0.6**	**0.1**	**0.6**	**20.9**	**0**	**0.1**	**0.7**	**10**	**0**	**0.1**	**0.1**	**4.2**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Temnopleuridae	0	1 7	0 2	0					0 7	10	0	0 1												
Heliocidaris erythrogramma	0 2	18 3	0 4	0 1	0 6	18 6	0 1	0 1																
POLYCHAETA	**5.7**	**74.2**	**8.4**	**4.9**	**0.7**	**34.9**	**2.5**	**0.6**	**0.1**	**3.3**	**17.9**	**0.8**	**0.2**	**4.2**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Eunice tentculata	0	3 3	0 1	0																				
Lysaretidae	0	1 7	0	0																				
Sabellidae					0 3	23 3	2 3	0 4																
Lumbrineridae	0 1	5 0	0 1	0																				
Nereidae	5 5	70	8 0	6 3	0 3	16 3	0 2	0																
Spinunculidae	0	1 7	0	0																				
Unknown polychaete	0 1	11 7	0 2	0					0 1	3 3	17.9	0.8	0 2	8 3	0 5	0 1								
ISOPODA	**3.9**	**80.6**	**10.4**	**6.6**	**12.8**	**80**	**19.2**	**14.9**	**2.8**	**36.7**	**0.5**	**2.1**	**1.0**	**41.7**	**1.3**	**0.8**	**0.4**	**2.4**	**0**	**0**	**0**	**0**	**0**	**0**
Zuzara venosa	3 9	80 6	10 4	6 6	12 8	100	19 2	14 9	2 8	36 7	0 5	2 1												
Shaeromatidae													1 0	41 7	0 7	1 1								
Idoteidae													0 1	4 2	0 6	0								
Arcturidae																	0 4	2 4	0	0				
PHAEOPHYTA	**0**	**0**	**0**	**0**	**0**	**4.7**	**0**	**0**	**0**	**0**	**0**	**0**	**0.3**	**20.8**	**0.0**	**0.1**	**3.4**	**9.8**	**0.2**	**0.2**	**21.7**	**84.6**	**26.3**	**25.4**
Sporochnales (Carpomitra costata)																					0 2	7 7	0 1	0
Dictyotales (Dictoya spp)																					7 0	23 1	7 3	4 3

Table 5.2 (cont.)

Prey Items	BMW % N	% FOO	% W	% IRI	BTR % N	% FOO	% W	% IRI	BTW % N	% FOO	% W	% IRI	PUW % N	% FOO	% W	% IRI	LSB % N	% FOO	% W	% IRI	MBL % N	% FOO	% W	% IRI
Fucales (Durvillaea potatorum/ Phyllospora comosa)																					4 1	5 8	5 7	0 8
Laminariales (Ecklonia radiata)																					0 6	3 8	0 3	0
Unknown brown algae																	3 4	9 8	0 1	0 2	4 3	26 9	4 1	2 9
RHODOPHYTA	**0**	**0**	**0**	**0**	**0.1**	**11.6**	**0**	**0**	**0**	**0**	**0**	**0**	**0.1**	**4.2**	**0.5**	**0.0**	**2.7**	**17.1**	**0**	**0.3**	**37.7**	**88.5**	**36.6**	**41.2**
Rhodymeniales (Rhodymenia australis/Champia viridis)																					6 4	61 5	5 7	8 4
Gigartinales (Callophyllis spp./Plocamium dilatum/Phacellocarpus pepperocarpus)																					17.8	73 0	23 7	16 7
Gelidiales (Pterocladia capillacea)																					0 6	3 8	0	0
Ceramiales (Ballia callitricha/Lenormandia marginate)																	1 3	9 8	0	0 1	3 7	15 3	2 0	0 3
Filamentous reds																					6 7	53 8	4.5	7.8
Unknown red algae																	1 3	7 3	0	0 1	7 2	19 2	10.9	4 5
ALGAE (OTHER)	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.1**	**4.2**	**0.2**	**0**	**0**	**0**	**0**	**0**	**26.4**	**76.9**	**24.4**	**24.4**
Ulvales (Enteromorpha spp and Ulva spp.)																					1 5	15 3	1 3	0 3
Chlorophyta													0 1	4 2	0 2	0								
Unidentified algal material																					24 9	73 1	23 1	24 1
OTHER	**4.6**	**83.9**	**4.5**	**4.3**	**5.6**	**86.0**	**1.7**	**1.6**	**5.5**	**26.7**	**31.2**	**1.6**	**1.3**	**37.5**	**13.0**	**4.2**	**0**	**0**	**0**	**0**	**14.0**	**53.8**	**12.8**	**9.0**
Ascidian	0	3 3	0	0																				
Bryozoan	0 1	6 7	0	0																	1 1	23 1	0 7	0 5
Ischnochiton cariosus	0 6	45 0	2 0	0 8	0 3	27 9	0 4	0 1	1 1	10	1 1	0 3	1 1	33 3	9 2	5 5								
Calanoid copepod	0 1	1 7	0	0																				
Cumacea	0 1	1 7	0	0	0.7	16 3	0	0 1																
Mysideacea	0 7	11 7	0 5	0 1	0 1	9 3	0	0	3 2	6 7	25 2	2 5												
Gymnangium superbum																					2 1	26.9	1 2	1 2
Ostracod	1 2	40	0 4	0 4	3 2	58 1	0 6	1.3	0 2	3 3	0	0												
Pallenopsis gippslandiae	0 3	28 3	0 1	0 1	0 3	14 0	0	0																
Sponge	0	1 7	1 1	0									0.1	4 2	3 9	0 3					10 8	30.8	0	4 3
Zeuxo spp	1 4	25 0	0 4	0 3	0 5	18.6	0 5	0 1																
Unknown ascidian	0 1	3 3	0	0																				
Stomatopoda					0 4	20 9	0	0																
Tanaeidae									0 2	3 3	0	0												
Nebaliacea					0 1	4 7	0	0																
Asteroida									0 1	3 3	0 3	0												
Tosia spp									0 4	3 3	0 1	0												
Coscinasterias muricata									0 3	3 3	4 5	0 2												
Nematode													0 2	8 3	48.1	6 5								

5.3.3 Ecosystem models based on dietary information

The detailed initial model (Fig. 5.4) was produced on the basis of dietary information obtained during this study and information on 10 different prey groups from the published literature (Table 5.1, n > 1710). For each simplified model, variables were aggregated following the production of dendrograms (Fig. 5.5) with the exception of detritus. Detritus was not aggregated with any variable in the simplified models, as it is functionally very different to all other model variables.

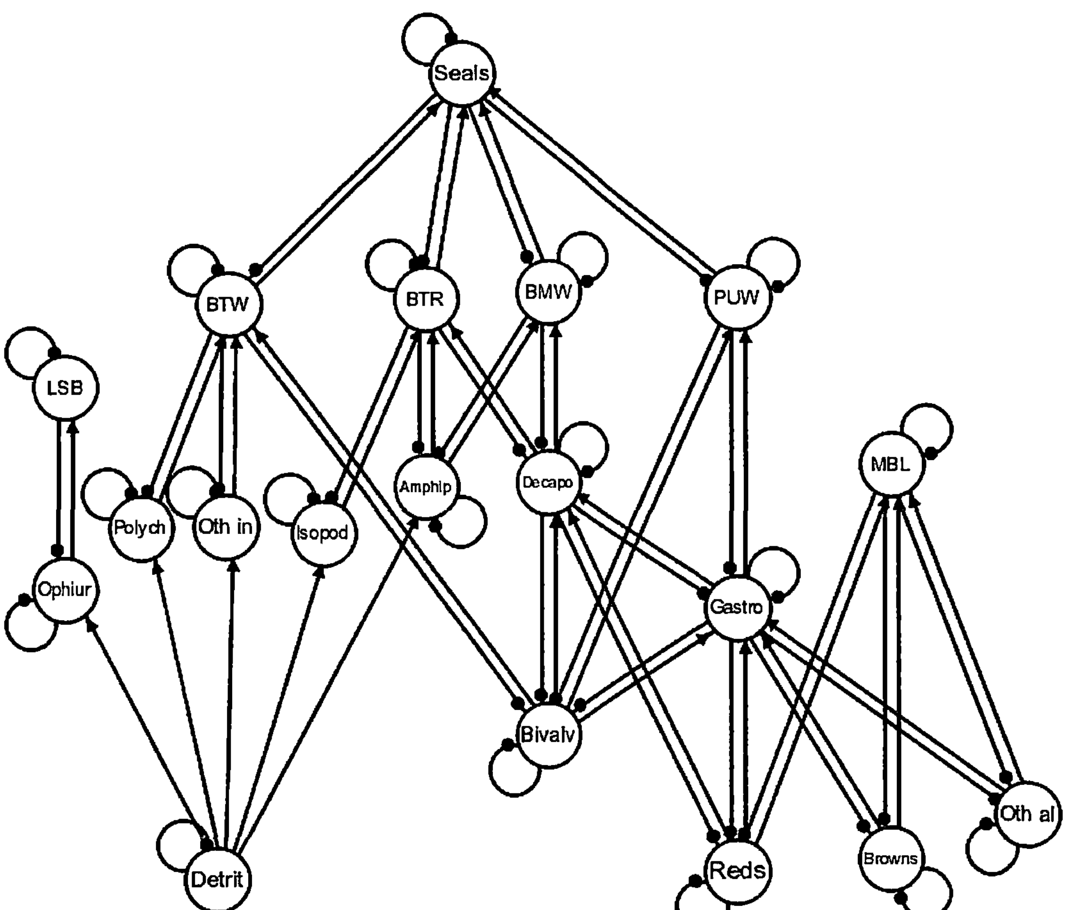


Figure 5.4 Initial model with fish abbreviations as in Figure 5.3. Other abbreviated variables are: bivalves (Bivalv); red algae (Reds); other algae (Oth al); brown algae (Browns); gastropods (Gastro); decapods (Decapo); amphipods (Amphip); ophiuroids (Ophiur); polychaetes (Polych); isopods (Isopod); other invertebrates (Oth in); and detritus (Detrit).

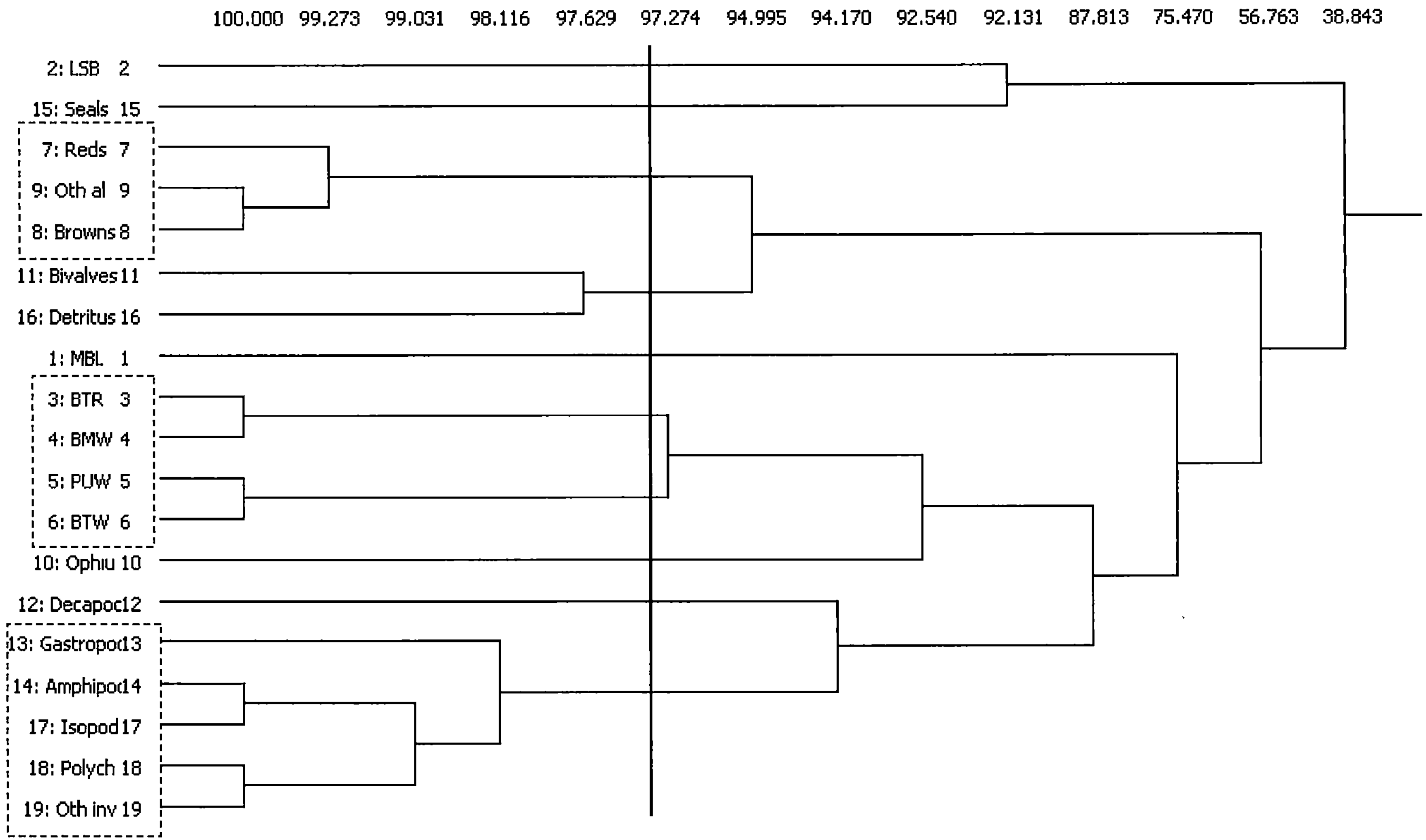


Figure 5.5 Dendrogram displaying the regular equivalence (similarity) between variables in the initial model. The dashed lines represent variables that were aggregated as a result of these analyses for the REGE model. The solid vertical line shows the cut-off point for aggregating variables. Variable names are in Figure 4. The numbers at the top refer to the level of similarity between variables.

In support of the results found using Morisita's Index, aggregation of the initial model by regular equivalence split benthic invertebrate feeding fish into two groups; one with bastard trumpeter and banded morwong and the other with purple wrasse and blue throat wrasse (Fig. 5.6a). In contrast, aggregation using Bray Curtis similarity and Euclidean distance resulted in the grouping of purple wrasse, blue throat wrasse, banded morwong and bastard trumpeter into a single benthic invertebrate-feeding group (Fig. 5.6b-c). The separation of benthic invertebrate feeding fish into two groups retained a greater level of detail on trophic linkages. For instance, the RE model showed bivalves were a prey of wrasse but not of bastard trumpeter and banded morwong. A number of conflicting predictions occurred between models as a result of different linkages between variables in each model. For instance, wrasse have a negative effect on bastard trumpeter and banded morwong in the RE model while positive effects are produced in the BC and ED models (Table 5.3).

A comparison of predictions between the simplified and detailed initial models revealed the RE model had the largest number of predictions consistent with the initial model (86 %) followed by the BC model (73 %) and the ED model (65 %). Examination of aggregation error allows for the discrimination of the 'best' prediction based on the initial model when conflicting predictions occur between the simplified models.

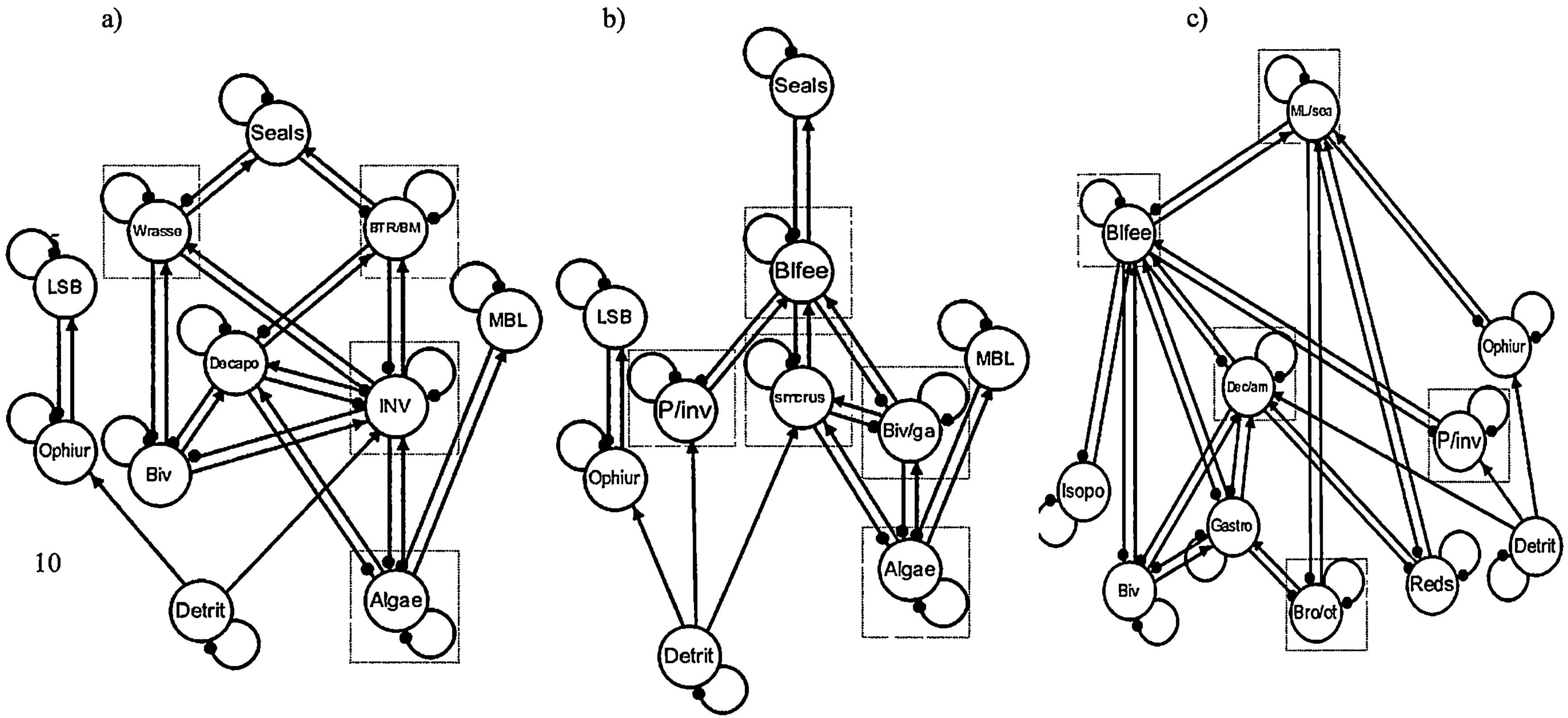


Figure 5.6 The a) RE, b) BC and c) ED models as simplified from the initial model in Figure 4.4. Boxes represent aggregated variables from Figure 4.4 following aggregation. Aggregated variable names are: bastard trumpeter and banded morwong (BTR/BM); purple wrasse and blue throat wrasse (Wrasse); banded morwong, bastard trumpeter, blue throat wrasse and purple wrasse (BI fee); isopods, polychaetes, other invertebrates, amphipods and gastropods (INV); red, brown and other algae (Algae); polychaetes and other invertebrates (P/inv); decapods, isopods and amphipods (smcrust); bivalves and gastropods (Biv/ga); Brown algae and other algae (Bro/ot); decapods and amphipods (Dec/am); and marblefish, lon-snouted boarfish and seals (ML/sea).

Table 5.3 Selected conflicting predictions of response to perturbation (increase to a variable) for models aggregated using regular equivalence (RE), Bray Curtis similarities (BC) and Euclidean distance (ED). Where variable names were not the same between models the equivalent name has been given. Effects are: negative (-); positive (+); no effect (0); and ambiguous (?) when the effect may be positive or negative.

RE model			**BC model**			**ED model**		
Variable increased	**Variable affected**	**Response**	**Variable increased**	**Variable affected**	**Response**	**Variable increased**	**Variable affected**	**Response**
1. Wrasse	Bastard trumpeter/ banded morwong	-	Benthic invert. feeders	Benthic invert. feeders	+	Benthic invert. feeders	Benthic invert. feeders	+
2. Wrasse	Bivalves	-	Benthic invert. feeders	Benthic invert feeders	?	Benthic invert. feeders	Bivalves	+
3. Bastard trumpeter/ banded morwong	Wrasse	-	Benthic invert. feeders	Benthic invert. feeders	+	Benthic invert. feeders	Benthic invert. feeders	+
4. Invertebrates (gastropods)	Bivalves	-	Bivalves/ Gastropods	Bivalves	+	Gastropods	Bivalves	-
5. Seals	Marblefish	-	Seals	Marblefish	-	Marblefish/long-snouted boarfish /seals	Marblefish	+
6. Ophiuroids	All variables excluding ophiuroids and long -snouted boarfish	0	Ophiuroids	All variables excluding ophiuroids and long -snouted boarfish	0	Ophiuroids	All variables excluding ophiuroids and long-snouted boarfish	Mixed effects (+ or -) except detritus (0)

5.4 DISCUSSION

Ecosystem models are becoming more widely used in conjunction with conventional stock assessment and to investigate fishery and ecosystem sustainability. The majority of ecosystem models are based on food webs (Fulton et al. 2003) and trophic information is therefore essential to facilitate their use in management. Detailed trophic information may result in complex models and incomprehensible results. Model simplification may be useful in such situations to investigate ecosystem structure and clarify results. When simplifying models through variable aggregation, aggregation error should be minimised to produce accurate predictions based on the complex model. The more widespread application of qualitative modelling and the aggregation methods used in this study may prove to be beneficial for general investigations into fishery and ecosystem sustainability.

Dietary analyses are often undertaken to provide information and reduce the uncertainty of ecosystem linkages and properties (Deb 1997). The diets of the study species had been investigated in few studies in the past (e.g. Fenton 1996, Bulman et al. 2001) and additional data collection was undertaken to ensure appropriate information on each species were available from eastern Tasmania. The sample sizes used in this study were not large. As a result, they were bolstered by the literature and prey were grouped into higher taxonomic orders for analysis. Furthermore, it is acknowledged that the identification of all trophic links is impossible (Deb 1997) and an adequate sample size was obtained to accurately identify the richness of prey items in the fish diets. The number and affect of weak links in the trophic web may be estimated using metrics, such as the Pareto *c* Index (Pinnegar et al. 2005), during quantitative analyses.

Simplification using regular equivalence (RE) produced the least aggregation error between the initial and simplified models. The dynamics of the RE model following perturbation may therefore be assumed to be a good representation of the initial model dynamics. The RE model supported the findings of the dietary analysis and Morisita's index by aggregating purple wrasse and blue throat wrasse, and bastard trumpeter and banded morwong. In contrast, both the BC and ED models aggregated all four benthic invertebrate feeding fish into one variable. This resulted in a number of different predictions between simplified models and highlighted the importance of model structure on predictions. The RE model suggested that increases to wrasse will have a negative effect on bastard trumpeter and banded morwong. This was due to a higher number of negative than positive feedback cycles between these variables and may indicate competition. This potential competition may also have implications for the commercial wrasse and banded morwong fisheries. For instance, substantial increases in the catch rate of wrasse may reduce the negative effect of wrasse and allow banded morwong to increase in abundance. The aggregation of these two fish variables in the BC and ED models would not have allowed for the determination of these negative effects. The prediction of competition between banded morwong and wrasse will be investigated in a quantitative ecosystem model (Chapter 6).

Bray Curtis similarities do not take the predators of the species into account during calculations. For example, in the BC model gastropods were aggregated with their bivalve prey. It has been suggested that predators and their prey should not be aggregated in many studies (O'Neill and Rust 1979, Gardner et al. 1982, Fulton et al. 2003) as this increases the error of the model. As a result, aggregating gastropods and bivalves affected model predictions. The BC model predicted a positive effect of gastropods on bivalves because they form part of the same variable and are assumed to increase in abundance simultaneously. In contrast, the RE and ED models retained bivalves and gastropods as separate variables and this resulted in negative effects on bivalves as a result of gastropod predation. Similarly, the simplification of the initial model using Euclidean distance resulted in the aggregation of seals with members of a lower trophic level; marblefish and long-snouted boarfish. Although seals were not included as direct predators on these fish species, aggregating members of different trophic levels creates error as shown with gastropods and bivalves. This aggregation may also create error as a consequence of variables with different production and turnover rates being placed in a single variable (O'Neill and Rust 1979, Gardner et al. 1982).

Bray Curtis and Euclidean distance measures do not take all 'neighbours' (i.e. predators and prey) into account when calculating similarities and this results in aggregation error. Long-snouted boarfish and ophiuroids formed a separate subsystem in the initial model as they were only connected to the rest of the system through a one-way link with detritus. Variables in separate subsystems only affect other variables in the subsystem and cannot be affected by alterations in the abundance of variables in the rest of the system. This detail was retained in both the RE model and BC models; however, the aggregation of seals with marblefish and long-snouted boarfish in the ED model connected ophiuroids to the rest of the system. This contributed to the high level of error in the ED model and occurred because Euclidean distance did not account for different prey types and linkages in the system.

As models become more important in resource management around the world, the problem of uncertainty in model prediction is increasing. In order to produce constructive and sustainable management regimes, ecosystem models need to be based on information specific to the study at hand. The results of the dietary analyses in this study allowed the production of alternative qualitative models from which aggregation error and model structure were investigated. It is important that techniques for model aggregation should be based on the similarity of all neighbours (e.g. predators and prey) as well as factors such as life history characteristics. In the past, regular equivalence has generally only been used in studying social networks (Luczkovich et al. 2002). Future studies should use the methods outlined in this study to simplify models, minimise aggregation error and increase model predictability prior to further ecosystem analyses.

5.5 ACKNOWLEDGEMENTS

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6. USE OF QUALTATIVE MODELS TO INFORM QUANTITTIVE ECOSYSTEM STUDIES: ROBUSTNESS OF CONCLUSIONS AND POTENTIAL IN DATA-POOR SITUATIONS

6.1 INTRODUCTION

Ecosystem models are an important tool in the management of marine systems, particularly as ecosystem-based fisheries management (Larkin 1996) and management strategy evaluation (MSE, Smith 1994) are becoming used with increasing frequency. The selection of an appropriate model is essential to the effectiveness of the study as each type of model, whether qualitative, statistical or mechanistic, possesses different combinations of the attributes that are important for modelling. These attributes are realism, generality and precision (Levins 1966, General Introduction). The use of multiple types of models may complement each other and increase the understanding of system dynamics by clarifying different aspects of a system.

Obtaining results from models with different uncertainties and assumptions may be beneficial as it allows investigation into the robustness of conclusions (Levins 1966, Wilson 1998, Sainsbury et al. 2000). The identification of robust conclusions may be beneficial for managers during risk assessment as an alternative measure of likelihood (Nagy et al. 2007). An understanding of modelling processes is necessary to allow the separation of robust conclusions from conclusions that are consistent between models but arise simply due to modelling techniques. In addition, this understanding of modelling processes can be useful when predictions between models differ to identify the specific aspects of the models or data that caused the discrepancy. The results from both qualitative and quantitative models were therefore assessed and compared in this chapter.

In addition to allowing investigations into the robustness of conclusions, the use of multiple types of models can be beneficial in data-poor systems. Quantitative ecosystem models produced in these systems are often criticised due to the potential for higher uncertainty and lower precision of model predictions as a result of limited data input (Latour et al. 2003, Plaganyi and Butterworth 2004). For this reason, the practical use of ecosystem models remains infrequent in the management of data-limited systems. Yet, precise estimates of the magnitude of response are not always necessary to inform management questions (Ramsey and Veltman 2005). For instance, Eisenack and Kropp (2001) used qualitative reasoning to aid the development of advice for management strategies to prevent the depletion of fish resources. Similarly, qualitative models have been used to provide advice to fisheries managers with regard to indicators of change (Hayes et al. 2008). The simplicity and generality of qualitative models determines that this technique may also be useful in focussing quantitative studies and data collection needs. Focussing quantitative studies can reduce the data requirements and may thus facilitate ecosystem investigations in data-poor situations.

Qualitative models have been used to aid the general understanding of ecosystem dynamics on Lihir Island, Papua New Guinea, as well as guide further research into fish populations, artisanal fishing and land use (Dambacher et al. 2007). In this chapter, an Ecopath with Ecosim (Walters et al. 1997) model was produced to investigate the predictions generated by the qualitative models (Chapters 3 and 5) and to investigate the robustness of conclusions between model types.

6.2 METHODS

6.2.1 Study ecosystem and fishery

Prior to the construction of models, the spatial scale of the ecosystem was defined using available community structure and commercial fishing data from around Tasmania (Chapter 2). The inshore reefs of eastern Tasmania were found to form a discrete ecosystem during these analyses (Chapter 2). The banded morwong (*Cheilodactylus spectabilis*) fishery was of particular interest due to its potential ecological impact in the ecosystem. In addition, the management of this fishery was under review, with output controls, specifically quota management and a total allowable catch (TAC) recently introduced (October 2008).

6.2.2 Qualitative modelling

Qualitative models were used to investigate the effects of catch controls (TAC), an increased occurrence of urchin barrens and a reduction in macroalgal biomass (Chapters 3 and 5). In addition, the importance of using ecosystem-specific trophic information was investigated and focussed on the diets of commercially important species in the banded morwong fishery. To provide some perspective on why focussing ecosystem analyses using qualitative models may be necessary, the number of potential predictions that could have been investigated in this study would be 1.37×10^9. This calculation is based on the 37 functional groups in the Ecopath model and is the sum of all combinations of change in the system minus the combination where no changes were made (2^{37}-1).

A number of topics were highlighted for investigation using Ecopath with Ecosim during qualitative modelling:

- The importance of links (potential competition) between banded morwong/bastard trumpeter and wrasse;
- The capacity for a TAC on the banded morwong fishery to allow overall biomass and catch biomass of banded morwong to recover;
- The capacity for rock lobster to limit the effects of urchin grazing and, as a result, increase the biomass of foliose algae;
- The role of increased urchin abundance with continued fishing on banded morwong, wrasse, and abalone fisheries;
- The effects of a reduction in foliose algal primary production (due to rising SST, a consequence of climate change) on the food chain and sustainable fisheries catch.

6.2.3 Qualitative predictions

The symbolic adjoint matrix was used in Chapter 3 to identify individual feedback cycles contributing to certain ambiguous predictions. Another method for investigating ambiguity is weighted predictions. Weighted predictions are the ratio of the net to total number of complementary terms (Chapter 3) contributing to a response and can be used to investigate the degree to which ambiguity occurs for each prediction (Dambacher et al. 2002). Values range between zero and one with values near zero being highly indeterminate. The closer the weighted prediction is to one, the greater the reliability of the prediction sign. If one positive and one negative complementary term exist for a given prediction, the probability of obtaining a positive (or negative) response is 0.5. Ambiguity may also occur to a lesser extent. For example, four negative terms and one positive term would produce a –3 in the adjoint matrix, as one negative term is cancelled by the single positive. Each element of the adjoint matrix may be weighted by the number of terms contributing to the response (Dambacher et al. 2002). For instance, in the previous example, the weighted prediction would be 3/5 = 0.60, which is the number of terms in the adjoint matrix (for that specific interaction) divided by the total number of complementary terms that contribute to the response. The weighted prediction provides an indication of the potential for sign indeterminacy, which can be particularly valuable for the comparison of qualitative model predictions. A matrix of weighted predictions (Appendix 6.1) for the trophic model (Chapter 5) has been included in this chapter as it has relevance for the assessment of results between this qualitative model and the Ecopath with Ecosim model.

6.2.4 Trophic information and functional groups

Thirty-seven functional groups were used in the Ecopath with Ecosim model and were defined based on abundance, similarity of trophic role (e.g. diet and life history similarities), importance to commercial fisheries and species or groups found to be important during qualitative modelling. For instance, bastard trumpeter (*Latridopsis forsteri*) and wrasse (*Notolabrus* spp.) were included in the model as separate variables because they are of commercial significance and were highlighted for further investigation in the qualitative models. Blue throat wrasse (*Notolabrus tetricus*) and purple wrasse (*N. fucicola*) were aggregated into a single variable, as the majority of commercial wrasse fishery records do not specify species. In addition, these species were found to have a relatively similar diet (Chapter 5). In contrast, juvenile and adult banded morwong have been found to have substantially different diets (S. J. Metcalf, unpublished data) and are subject to fishing pressure only as adults due to legal size limits. This species was therefore separated into juveniles and adults using the multi-stanza group function where one variable has two components that are linked through reproduction and maturation. No other species were split using this function; instead fish functional groups that comprised more than one species (i.e. omnivorous fish and planktivorous fish) were separated into different groups according to size. This was undertaken due to differing production and consumption rates between large and small fish (Polis 1984). Fish in the large size class were ≥50 cm fork length (FL) and fish in the smaller size class were <50 cm FL (Edgar 2000). The model consisted of: 13 fish; 14 invertebrate; two shark and rays; one marine mammal; one seabird; one detrital; and five primary producer groups.

Dietary information for banded morwong, bastard trumpeter, blue throat wrasse, purple wrasse, long-snouted boarfish and marblefish (*Aplodactylus arctidens*) was collected within the ecosystem and analysed in Chapter 5. All other dietary data was obtained from published studies (see Appendices 6.2 and 6.3) within Tasmania or, if unavailable, from similar rocky reef habitats in southeastern Australia or New Zealand.

6.2.5 Ecopath with Ecosim model

Ecopath with Ecosim (Version 5) was used to investigate the trophodynamics and effect of fisheries harvest on the inshore reefs of eastern Tasmania. This model analysed a system of differential equations to estimate trophic flow between n functional groups. Equations follow the form

$$B_i.(\frac{P_i}{B_i}).EE_i - \sum_{j=1}^{n} B_j(\frac{Q}{B})_j.DC_{ji} - EX_i = 0 \qquad (i = 1 \ldots n) \qquad (6.1)$$

where: B_i is the biomass of group i; $\frac{P_i}{B_i}$ is the production-biomass ratio used as an estimate of total natural mortality under steady-state conditions; EE_i is the ecotrophic efficiency; B_j is the biomass of predator j; $(\frac{Q}{B})_j$ is the consumption-biomass ratio of predator j; DC_{ji} is the fraction of prey i in the diet of predator j; and EX_i is the export of group i including fisheries catches (Walters et al. 1997, Manickchand-Heilman et al. 2004).

Commercial fisheries catch data was used to provide an estimate of exploitation rates within the ecosystem. Fishery data was available from 1994 to 2005/06 at the time of model production and was obtained from multiple sources (Table 6.1). The proportion of non-target species removed or expected to die following capture and discarding in the banded morwong fishery (Chapter 4) were included to provide a more accurate assessment of the ecosystem effects of fishing. Where possible, data on non-target species retained in each fishery was also included in the model.

Table 6.1 References used in the Ecopath model for commercial fisheries catch data within the inshore reef ecosystem of eastern Tasmania. Fisheries are grouped as they were used in the model.

Fishery	**Reference**
Banded morwong fishery	Ziegler et al. 2008
Wrasse fishery	Ziegler et al. 2008
Rock lobster fishery	Haddon and Gardner 2008
Abalone fishery	Tarbath et al. 2007
Other finfish and cephalopod fisheries	Ziegler et al. 2008

Estimates of fish, invertebrate and algal abundance were obtained from biodiversity surveys for the years 1994 to 2006 on the east coast of Tasmania (G. Edgar and N. Barrett, pers. comm.). Biomass was estimated through the multiplication of the average weight, obtained from published literature or information gained directly from researchers, with the number of individuals sighted per metre of the area surveyed of investigated (Appendix 6.4). Estimates of biomass for species that were not recorded in the biodiversity surveys (i.e. seals, seabirds) were obtained from the literature or directly from researchers (Appendix 6.4). Production/biomass, consumption/biomass and ecotrophic efficiency were derived primarily from the scientific literature (Appendix 6.4) and occasionally from FishBase (www.fishbase.org). Individual species parameter values were weighted by species biomass within the ecosystem if they were to be included in aggregated groups (sensu Okey et al. 2004b).

The Ecopath model was balanced iteratively by altering the diet matrix as this data was suggested to have the least certainty of all data included in Ecopath models (Okey et al. 2004a). Groups with the highest EE were altered first. Small changes were occasionally made to P/B and B; however, changes to values based on field data, such as most biomass estimates, were avoided.

Model parameter uncertainty was addressed through the use of the Ecopath sensitivity routine (Christensen et al. 2005). In this routine, the sensitivities of model-estimated ecotrophic efficiencies (EE) were investigated by sequentially changing the input parameters, biomass (B) and consumption/biomass (Q/B) in steps of 10% between -50% and 50%. The resultant change in EE was averaged across all impacted groups (Bulman et al. 2006) and provided an estimate of the degree to which the model was driven by each parameter.

Niche overlap was used to investigate the potential for competition between banded morwong, bastard trumpeter and wrasse. Overlap was assumed to be significant if greater than 0.6 (Wallace and Ramsey 1983). Mixed trophic impacts (Ulanowicz and

Puccia 1990) were calculated using Ecopath and used to assess the trophic impact of individual functional groups. The mixed trophic impacts were then summed for each functional group to calculate the total impact per group within the ecosystem. Mixed trophic impacts are related to qualitative models through the use of the inverse of the negative community matrix ($-\mathbf{A}^{-1}$) (Ulanowicz and Puccia 1990). The inverse of the negative community matrix is used in the calculation of predictions of response to perturbation in both qualitative modelling and mixed trophic impacts. Yet, the two methods differ in that mixed trophic impacts use net quantitative estimates of flow (e.g. gC m^{-2} $year^{-1}$, Bondavalli and Ulanowicz 1999) between variables, while qualitative modelling uses the qualitative effects (-1, 1, 0) between variables from the community matrix. If model structure is the same when using mixed trophic impacts and qualitative modelling, and if no ambiguity of flows or predictions occur (i.e. only positive or negative impacts between two variables), the results obtained through the different methods will be the same. In contrast, if any structural differences or ambiguity do occur, dissimilar predictions may emerge between the two techniques. For example, in mixed trophic impacts one strong positive (or negative) flow could effectively cancel multiple weak negative (or positive) flows. The predicted response of the variable in question may therefore be positive. When using qualitative modelling, the presence of both positive and negative links between variables would create ambiguity, yet the prediction sign would be determined by the net number of positive and negative signs contributing to the response, not their magnitude. For the previous example, the qualitative prediction would be negative.

6.2.6 Simulations and investigation into qualitative modelling predictions

Historical fishery time series were used to force fisheries catches and to provide an estimate of model fit using Ecosim. In addition, satellite-derived chlorophyll measurements were obtained from the Sea-viewing Wide Field-of-view Sensor (SeaWiFS). The Hoyo estimate (Howard and Yoder 1997) of primary productivity was used as a proxy for phytoplankton biomass to force the Ecosim simulations.

Various simulations and statistics were used to investigate the predictions produced during qualitative modelling (Table 6.2). The impact of perturbations to the overall biomass (t/km^2) of selected functional groups and the catch rates of fisheries were investigated. The change in biomass and catch biomass (tonnes captured/km^2/yr) that occurred following each perturbation was assessed. To clarify, catch rates refer to the rate of capture by the fishery (i.e. the frequency of capture at a certain level of effort) and signify the level of effort being used while the catch biomass refers to the tonnes of fish captured as a result of fishing effort. All simulations were run for 20 years. Simulated changes of $\pm 25\%$ of overall biomass or catch, based on 2005 levels, were used to investigate each prediction. This level of change was selected as Christensen (1998) cautioned Ecosim users against extremely large changes in fishing pressure and this level of change was thought to be sufficient to observe a range of ecosystem responses. As the abalone and rock lobster commercial fisheries are subject to TACs in the inshore reef ecosystem, TACs were included for these fisheries in all simulations by limiting catch to 2005 levels. The wrasse and 'other' fisheries are not subject to TACs and 1995-2005 fishing rates were replicated for the remainder of the 20 year simulation to represent typical catch rates.

Using simulations, the relationship between wrasse and banded morwong/bastard trumpeter was investigated by altering wrasse and banded morwong (includes catch of non-target bastard trumpeter) fishing rates separately. This was undertaken in order to reduce their abundance and to observe any reciprocal change in abundance in the unaltered group. Investigation into the impact of a TAC on the overall biomass and catch biomass of banded morwong was undertaken by retaining fishing rates at 2005 levels for the remainder of the simulation. Following the investigation into a TAC, fishing rates were reduced to 75% of the 2005 rate to investigate the impact of a reduced TAC. In addition, a number of simulations were undertaken to investigate the impact of urchins, primary production (foliose algae and phytoplankton) and rock lobster on other groups in the system and the catch biomass of each fishery. Each simulation used a 25% decrease in biomass or fishing rate to investigate these impacts (Table 6.2). Simulations in which primary production was altered were undertaken to investigate the potential effects of climate change on the inshore reef ecosystem (Hobday et al. 2006). While phytoplankton was not included in the qualitative models (Chapters 3 and 5) it was included in the analyses using Ecopath with Ecosim as they are an important source of primary production on the Tasmanian coast (Harris et al. 1987). Simulations altering phytoplankton separately and simultaneously with macroalgae were undertaken to investigate the impact of a decrease in both sources of primary production in the ecosystem.

Table 6.2 Methods used to compare predictions generated by qualitative modelling. Further information on predictions is provided in Chapters 3 and 5. NA- not applicable.

Qualitative predictions	Ecopath statistics used	Simulations used
1) An increase in the biomass of banded morwong/ bastard trumpeter will decrease wrasse biomass and vice versa	• Niche overlap (predators and prey) • Mixed trophic impacts	• 25% increase in banded morwong fishing rates • 25% increase in wrasse fishing rates
2) A TAC on the banded morwong fishery will allow banded morwong biomass and catch biomass to increase	NA	• Banded morwong fishing rates were held constant at 2005 fishing rates • Banded morwong fishing rate was decreased by 25% while all other fishing rates remained constant at 2005 levels (as a TAC at 2005 levels did not allow banded morwong biomass to increase)
3i) Rock lobster limits urchin abundance and, as a result, can increase the biomass of foliose algae	NA	• Rock lobster fishing rate was decreased by 25%
3ii) Increased urchin abundance with continued fishing has a negative effect on the catch biomass of the banded morwong, wrasse and abalone fisheries	NA	• Urchin grazing was increased by 25%
4) A reduction in primary production (climate change) will create negative effects up the food chain and reduce catch in all fisheries	• Mixed trophic impacts	• 25% reduction in biomass of all macroalgal functional groups • 25% reduction in phytoplankton biomass • 25% reduction in the biomass of all macroalgal groups and phytoplankton simultaneously

6.3 RESULTS

6.3.1 Ecopath results and statistics

All biomass changes in the sensitivity analysis were below 50% suggesting the model parameters were not overly sensitive to change (Bulman et al. 2006). Table 6.3 includes the results due to -50% and 50% changes in B and Q/B alone, as these would be expected to produce the greatest changes in EE. The largest changes occurred due to modifications to biomass and were observed in long-snouted boarfish, pelagic sharks and herbivorous gastropods. These results suggest these functional groups were relatively sensitive to changes in biomass; however, large changes in biomass were not undertaken for these species during model balancing.

Table 6.3 Results of the sensitivity analysis of model-estimated ecotrophic efficiency (EE) to ±50% variations in the input parameters biomass (B) and consumption/biomass (Q/B). The percent change is averaged across all impacted groups (N) and the sign (+, -) indicates the direction of the change. Indirect effects may alter the level of change in some groups (i.e. the positive and negative values are not equal).

Group impacted/parameter changed	**B**		**N**	**Q/B**		**N**
	-50% Change in EE	**50% Change in EE**		**-50% Change in EE**	**50% Change in EE**	
Pelagic sharks	-27.85	27.85	2	-27.85	27.85	2
Marine mammals	-14.54	7.88	5	-18.17	18.17	4
Seabirds	-9.77	1.35	6	-8.28	41.40	5
Small sharks + rays	-16.11	5.78	9	-10.68	10.67	8
Large omnivorous fish	-2.58	4.09	10	-8.24	8.24	9
Small omnivorous fish	22.15	-5.47	4	-3.80	3.80	3
Large piscivorous fish	-6.26	10.92	12	-13.92	13.92	12
Small piscivorous fish	-9.78	1.30	5	-8.54	8.52	5
Large planktivorous fish	8.61	-1.93	6	-7.90	9.66	5
Small planktivorous fish	-6.26	2.07	8	-	-	-
Large herbivorous fish	-18.87	3.37	3	-	-	-
Small herbivorous fish	-2.34	1.97	7	-	-	-
Adult banded morwong	46.62	-13.27	4	-6.75	6.75	2
Bastard trumpeter	29.53	-7.30	3	-5.70	5.70	2
Wrasse	21.35	-4.67	3	-4.87	4.87	3
Long-snouted boarfish	47.55	-14.20	4	-4.90	4.90	2
Cephalopods	-2.07	3.79	2	-7.33	7.33	10
Rock lobster	-5.91	3.61	10	-9.77	9.77	6
Decapods	-3.53	4.76	7	-10.19	10.19	7
Other filter feeders	-14.83	1.85	4	-13.57	13.57	3
Bivalves	-21.93	0.30	3	-17.10	17.10	2
Centrostephanus rodgersii	-10.96	2.38	5	-11.30	11.30	4
Echinoids	-8.23	1.30	7	-7.07	7.07	6
Asteroids	-8.70	8.7	1	-8.28	8.28	5
Carnivorous gastropods	-9.76	1.35	5	-20.00	20.00	5
Polychaetes and detritivores	-0.383	11.23	6	-37.60	37.60	1
Herbivorous gastropods	-31.20	2.15	2	-	-	-
Abalone	27.23	-5.00	3	-	-	-

Investigation into niche overlap showed banded morwong (adult) and wrasses had significant prey (0.878) and predator overlap (0.851). Juvenile banded morwong had significant predator overlap (0.898) with bastard trumpeter while adult banded morwong had significant prey overlap with this species (0.723). In addition, bastard trumpeter was found to have significant prey overlap with wrasse (0.672). All other pairwise comparisons of niche overlap between these species were non-significant.

Investigation into mixed trophic impacts found the majority of large impacts were due to top predators or fisheries. For example, pelagic sharks had a trophic impact of -0.864 on marine mammals, and the banded morwong fishery had an impact of -0.487 on adult banded morwong. Banded morwong, bastard trumpeter and wrasses had little impact on each other (<0.05). Total mixed trophic impacts (Fig. 6.1) were calculated to show the overall impact of each functional group on the ecosystem. The greatest impacts were due to detritus, phytoplankton, brown and red macroalgae.

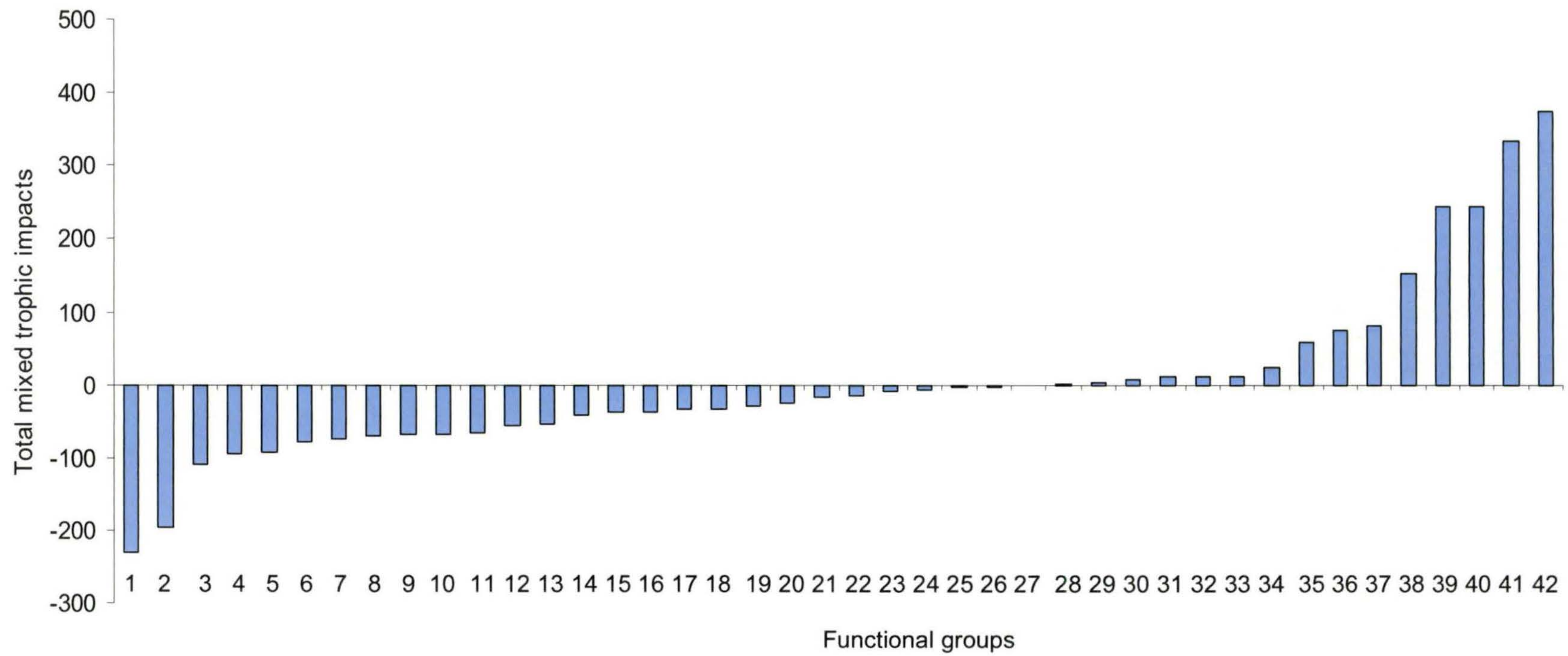


Figure 6.1 Total mixed trophic impacts of each functional group on the remaining groups. 1. Carnivorous gastropods, 2. large piscivorous fish, 3. small sharks and rays, 4. cephalopods, 5. asteroids, 6. banded morwong fishery, 7. wrasse fishery, 8. small piscivorous fish, 9. large omnivorous fish, 10. marine mammals, 11. decapods, 12. rock lobster, 13. abalone fishery, 14. small omnivorous fish, 15. bastard trumpeter, 16. rock lobster fishery, 17. other finfish fisheries, 18. pelagic sharks, 19. adult banded morwong, 20. seabirds, 21. herbivorous gastropods, 22. long-snouted boarfish, 23. wrasse, 24. juvenile banded morwong, 25. large herbivorous fish, 26. small herbivorous fish, 27. small planktivorous fish, 28. *Centrostephanus rodgersii*, 29. polychaetes and detritivores, 30. abalone, 31. bivalves, 32. echinoids, 33. large planktivorous fish, 34. crustose algae, 35. other filter feeders, 36. zooplankton, 37. green algae, 38. small crustaceans, 39. red algae, 40. brown algae, 41. phytoplankton, 42. detritus.

6.3.2 Simulations

Increased banded morwong and wrasse fishing rates

Increasing the fishing rates of banded morwong and wrasse through simulations was undertaken to investigate the potential for competition between these two variables (Table 6.4). The catch biomass (tonnes captured/km^2/yr) of both species gradually decreased until the end of each simulation. A 23% reduction in the catch biomass of banded morwong and a 12% reduction in the catch biomass of wrasse were observed following a simulated 25% increase in wrasse fishing rate (relative to 2005 catch rates). Similarly, increasing banded morwong fishing rate by 25% resulted in a 35% and an 8% decrease in the catch biomass of banded morwong and wrasse respectively.

Total Allowable Catch

Imposing a TAC on the banded morwong fishery by maintaining fishing rates at 2005 levels resulted in increases in the overall biomass of all functional groups excluding banded morwong adults (-28%) and juveniles (-10%), herbivorous fish (small -2%, large -32%), large planktivorous fish (-17%) and small piscivorous fish (-38%). In addition, a decrease in the catch biomass of banded morwong (-19%) and abalone (-2%) was observed during this simulation while the catch biomass of all other fisheries (i.e. wrasse and rock lobster fisheries) increased between 6 and 13%. As banded morwong biomass did not increase following the instigation of a TAC at 2005 levels, a reduced TAC (75% of 2005 rates) was investigated for this fishery. Both adult (2%) and juvenile (4%) banded morwong biomasses increased following this reduction in fishing rate. All other groups remained relatively unaffected from the previous simulation with the TAC set at 2005 catch levels.

Altered primary production

Simultaneously reducing phytoplankton and macroalgal biomass by 25% reduced the overall biomass of all groups and the catch biomass of all fisheries within the ecosystem. In contrast, the simulation decreasing all macroalgal groups by 25%, while phytoplankton biomass remained stable, resulted in decreases in the biomass of 40% of groups in the ecosystem. The largest decreases occurred in herbivorous gastropods and small herbivorous fish (both 47%). Echinoids, *C. rodgersii* and abalone also experienced decreased biomass at 32%, 37% and 27% respectively. Decreased catch occurred in all fisheries excluding 'other' fisheries. Large increases in biomass occurred in other groups that did not rely on macroalgae as a primary source of nutrients, such as small planktivorous fish (20%) and small crustaceans (19%).

In order to separate the impact of phytoplankton and macroalgae in the system, phytoplankton biomass was reduced by 25% while macroalgal biomass remained stable. This change caused a larger impact than a reduction in macroalgae with decreases to the biomass of 79% of groups in the ecosystem. Furthermore, catch biomass decreased in all fisheries by between 7% and 49%, with the largest decreases in the wrasse and banded morwong fisheries. Long-snouted boarfish had the greatest decrease in biomass of all ecosystem members at 58%. Abalone (13%) and echinoids (7%) had the highest increases in biomass following the reduction in phytoplankton.

Increased urchin biomass

A 25% increase to urchin biomass resulted in decreases in red (10%) and brown macroalgae (8%). Crustose algal and green macroalgal biomasses were both reduced by

3%. The catch biomass of all fisheries, excluding 'other' finfish (4%), decreased, with the largest change in the banded morwong fishery (43%) followed by the wrasse (22%), rock lobster (20%) and abalone (10%) fisheries. In contrast, an increase in biomass occurred in decapods (15%).

Reduced rock lobster fishing

A 25% reduction in rock lobster fishing rate resulted in a 10% increase in the catch biomass of the rock lobster fishery at the end of the simulation while the catch biomass of all other fisheries decreased. In comparison to the previous simulation, which focussed on the impact of increased urchin grazing, the decline in macroalgal biomass was reduced (6% for red algae and 8% for brown algae). The biomass of *C. rodgersii* decreased by 7% and a small decrease in echinoids was observed (2%). The biomass of abalone decreased 10% due to increased predation by rock lobsters.

Table 6.5 summarises the results of qualitative and quantitative models with regard to the predictions generated during qualitative modelling and any differences between the two types of models.

Table 6.4 Summary of major results from Ecosim simulations after 20 years.

Simulation	Change in overall biomass	Change in catch biomass
• Increased banded morwong fishing	↓ banded morwong (adults and juveniles), herb. fish (small and large), large planktivorous fish and small piscivorous fish ↑ all other functional groups	↓ banded morwong, ↓ wrasse
• Banded morwong TAC (at 2005 fishing rate)	↓ banded morwong (adults and juveniles), herb. fish (small and large), large planktivorous fish and small piscivorous fish ↑ all other functional groups	↓ banded morwong, abalone ↑ all other fished stocks
• Banded morwong TAC (at 75% of 2005 fishing rate)	↑ banded morwong (adults and juveniles)	↓ banded morwong
• Reduced phytoplankton and macroalgal (all groups) biomass	↓ all functional groups	↓ all fished stocks
• Reduced macroalgal biomass (all groups)	↑ in functional groups not reliant on macroalgae as a food source ↓ in all other functional groups (40% of groups)	↓ all fished stocks excluding 'other' fisheries
• Reduced phytoplankton biomass	↓ in 79% of functional groups	↓ all fished stocks
• Increased urchin biomass	↓ all macroalgal groups and crustose algae ↑ decapods	↓ all fished stocks excluding 'other' fisheries
• Reduced rock lobster fishing rate	↓ red and brown macroalgae, *C. rodgersii*, echinoids and abalone	↑ rock lobster ↓ all other fished stocks

Table 6.5 Summary of qualitative and Ecopath with Ecosim (EwE) model results by prediction. Not applicable is NA, total allowable catch is TAC and mixed trophic impacts is MTI. Foliose algae refers to brown, red and green macroalgae.

1) Qualitative predictions	**2) Reason/s for occurrence in qualitative model (Chapters 3 and 5)**	**3) Occur in EwE model? (Letters refer to different simulations or results)**	**4) Figures or results used as evidence**	**5) Reason/s result did not occur in EwE model (Letters correspond with column 3)**	**6) Reason for consistent results between models (Letters correspond with column 3)**
1) An increase in the biomass of banded morwong/ bastard trumpeter will decrease wrasse biomass and vice versa	Due to the dietary information collected in Chapter 5. More negative complementary terms (60) (between banded morwong/bastard trumpeter and wrasse) than positive complementary terms (24) determines the net number of terms is negative (-60 + 24 = -36), with a prediction weight of 0.42 (36 / 84 =0.42)	A) Mixed trophic impact- No B) Niche overlap- Yes C) Simulations- No	NA	A) Positive flows (i.e. banded morwong positively impacting prey of wrasse- bivalves) were stronger than the negative flows (i.e. banded morwong reducing the abundance of invertebrates for wrasse to prey on) in the calculation of MTI. C) MTI influenced the simulation results and determined the predicted changes were not observed.	B) The same dietary information (Chapter 5) was used for these fish species. Additional detail was used in the EwE diet matrix in comparison to the qualitative model; however, the niche overlap remained significant.
2) A TAC on the banded morwong fishery will allow banded morwong biomass and catch biomass to increase	Reduction in catch (C) due to management (Mngt) positively impacts biomass (B)	A) 2005 fishing rates-No B) 75% of 2005 rates- Yes	A) Management restrictions allow stock biomass to rebuild prior to the next fishing season. This occurs through the paths: Mngt –E – B and Mngt –C –P – E –B.	A) TAC set at 2005 levels too high for biomass to increase	B) TAC set at a level that allowed biomass to increase

Table 6.5 (cont.)

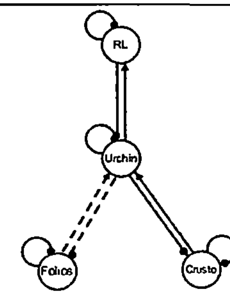
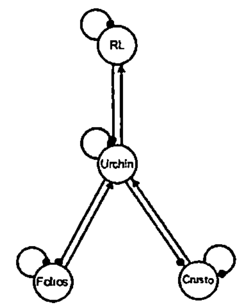
1) Qualitative predictions	2) Reason/s for occurrence in qualitative model (Chapters 3 and 5)	3) Occur in EwE model? (Letters refer to different simulations or results)	4) Figures or results used as evidence	5) Reason/s result did not occur in EwE model (Letters correspond with column 3)	6) Reason for consistent results between models (Letters correspond with column 3)
3i) Rock lobster limits urchin abundance and, as a result, can increase the biomass of foliose algae	Replenishment of foliose algae occurs when rock lobster allows foliose algae to increase in abundance (N^*) from near-zero $N^*_{\text{Foliose}} \geq 0$ Following the increase in foliose algal abundance N^* is substantially greater than zero ($N^*_{\text{Foliose}} > 0$)	Yes	 	NA	Trophic links between rock lobster, urchins and foliose algae in the diet matrix were strong enough to support the prediction
3ii) Increased urchin abundance with continued fishing has a negative effect on the catch biomass of the banded morwong, wrasse and abalone fisheries	Trophic-, habitat- and fishing-related linkages created negative responses in banded morwong, wrasse and abalone fishery catch biomass.	Yes	NA	NA	Trophic-, habitat- and fishing-related linkages resulted in a decline in the catch biomass of the banded morwong, wrasse and abalone fisheries. For example, less food was available for abalone.

Table 6.5 (cont.)

1) Qualitative predictions	2) Reason/s for occurrence in qualitative model (Chapters 3 and 5)	3) Occur in EwE model? (Letters refer to different simulations or results)	4) Figures or results used as evidence	5) Reason/s result did not occur in EwE model (Letters correspond with column 3)	6) Reason for consistent results between models (Letters correspond with column 3)
4) A reduction in primary production (climate change) will create negative effects up the food chain and reduce catch in all fisheries	Foliose algae is a main source of primary production and a reduced abundance of this algae will decrease the abundance of prey up the food chain (including fisheries). Phytoplankton was not included in this model.	Yes	NA	NA	Trophic links included in the diet matrix influenced the EwE model results and determined that the loss of a major source of primary production would have negative effects up the food chain including fisheries.

6.4 DISCUSSION

Conclusions were found to be generally consistent between qualitative and quantitative models, which suggests they are robust to the assumptions and limitations of the two modelling techniques employed (Laskey 1996). In both qualitative and quantitative models, a TAC was observed to increase the biomass of banded morwong. The Ecopath with Ecosim model was able to take this analysis one step further than the qualitative model, predicting that a TAC set at the 2005 fishing rate was too high to allow banded morwong biomass to increase while a small increase was predicted with a reduced TAC (75% of 2005 fishing rate). In addition, a reduction in primary production was predicted to create negative effects that were transmitted up the food chain to reduce the catch biomass of fisheries in both types of models. Similarly, both qualitative and quantitative models suggested increased abundances of rock lobster would increase algal abundance as a result of trophic linkages. Yet, the consideration of the robustness of conclusions from a small number of models must be undertaken with caution. Consistency between model outputs could occur as a result of being based on similar assumptions (e.g. relationships between variables or functional groups). In addition, structural and parameter uncertainty remain important issues in the use of model outputs for management (Chen and Ma 2006, Hosack et al. 2008). For these reasons, it is critical to have a detailed understanding of modelling processes to allow the separation of robust conclusions from those that are consistent simply due to modelling processes or input data.

Both the qualitative models produced in this study and Ecopath with Ecosim rely heavily on the trophic linkages included in the system. These trophic linkages were found to be the reason behind a number of the similarities in model results. This is not necessarily a problem and may result in a reasonable representation of ecosystem dynamics; however, a reliance on trophodynamics has been acknowledged as a potential issue in the Ecopath suite of models (Walters et al. 2000). Non-trophic models may be useful in this situation to provide a different perspective on ecosystem dynamics and an alternative model for the assessment of results. The use of such a model may help to overcome or identify any biases created. Another potential issue with modelling results is structural uncertainty because predictions are only as certain as the model structure from which they were calculated. For this reason, it is necessary to use the best available data and information when constructing ecosystem models and acknowledge that different model structures may exist (e.g. the urchin barren models with and without large rock lobster). While many potential issues and areas of uncertainty exist with regard to the ecosystem models presented in this case study and elsewhere, the two modelling techniques did differ substantially in both the number of variables and the detail of model output. The input data was also of the highest quality available. This provides weight to the predictions that were consistent between qualitative and quantitative models. These results can be used to inform management through processes such as a risk assessment for EBFM (Nagy et al. 2007) and to prioritise research in the future.

Similar to consistent results, consideration must be given to results that are inconsistent between modelling techniques. Inconsistent results can be useful when they highlight sources of uncertainty, such as different causal mechanisms (House et al. 2003). For example, the potential for resource competition between wrasse, bastard trumpeter and banded morwong was supported by qualitative modelling but contradicted in two of three investigations (simulations and mixed trophic impacts) using Ecopath with Ecosim. The potential for competition was observed during qualitative modelling due to a higher number of negative feedback loops than positive feedback loops between banded morwong and wrasse. This result was not without ambiguity, with 60 negative feedback cycles and 24 positive feedback cycles, between banded morwong/bastard trumpeter and wrasse. The analysis of the weighted predictions matrix (Dambacher et al. 2002) highlighted the relatively low weighted predictions associated with the predictions of change between these species. An increase in wrasse was predicted to have a negative impact on banded morwong/bastard trumpeter with a weighted prediction of 0.43 (App. 6.1). Similarly, an increase in banded morwong/bastard trumpeter was predicted to have a negative impact on wrasse with a weighted prediction of 0.14. These weighted predictions determine that only 43% and 14% of cycles contribute to the net direction of the response (Dambacher et al. 2002). In other words, few cycles contribute to the prediction sign obtained in the adjoint matrix. A minor quantitative change to one cycle may therefore alter the direction of the predictions. This corresponds with the results of the mixed trophic impacts, where no impact was observed between wrasse and banded morwong/bastard trumpeter. For example, a positive impact between banded morwong and wrasse may have occurred due to the predation of decapods. This interaction would be positive because a reduction in decapods would allow the prey of wrasse (bivalves) to increase in abundance, therefore benefiting wrasse. No impact between these fish species may have occurred if positive impacts, such as this, cancelled multiple weaker negative impacts. Similar results were found by Ulanowicz and Puccia (1990) where American alligators (*Alligator mississippiensis*) positively impacted populations of their prey, crayfish (*Procambrus alleni*). This occurred because alligators also feed on turtles (*Chelydra serpentina*), another predator of crayfish. The negative impact of alligators on crayfish (due to predation) was overwhelmed in the calculation of trophic impacts by the larger positive impact caused by the reduction in turtle predation on crayfish. Inconsistent results, in this way, can be important as they encourage further investigation into the mechanisms responsible for the differences. This additional investigation, in turn, may provide a greater understanding of model behaviour and system dynamics.

The assessment of perturbations using Ecopath with Ecosim provided new insights into the dynamics of the inshore reef ecosystem as well as management strategies that may be necessary to ensure fishery and ecosystem sustainability. For example, the Ecosim simulations suggested a TAC set at 2005 levels (48t, Ziegler et al. 2008) for banded morwong may be too high to sustain stock biomass based on the biomass estimates input into the model. This has important consequences for the management of the fishery, particularly if biomass estimates are assumed to be relatively accurate. A reduction of the TAC to 75% of 2005 levels (38t) allowed a small increase in the biomass of banded morwong. These results support a stock assessment model by Ziegler et al. (2008) that predicted a TAC of 40t would cause declines in banded morwong mature biomass and catch rates with almost 100% probability. This chapter, in addition to the results of Ziegler et al. (2008), has highlighted the need for a more conservative TAC in order to ensure banded morwong populations are sustainable. In addition, further investigation into the impact of fishing banded morwong should be undertaken. Investigating changes to primary production using Ecopath with Ecosim was important as alterations in macroalgae and phytoplankton on the inshore reef ecosystem may occur as a result of climate change (Eddyvane 2003, Hobday et al. 2006). The predicted decline in biomass and catch biomass for the majority of functional groups in the ecosystem suggests large reductions in fishing catch rates may be critical to ensure fishing remains sustainable. While this simulation used a 25% decline in primary production, even small reductions may require the adjustment of sustainable fishing rates by management. Monitoring of changes in primary production within the ecosystem may aid the prediction of sustainable harvest rates into the future.

Similar to a decrease in primary production, the simulation investigating an increase in urchin biomass predicted a decline in the biomass of a large number of functional groups as well as reductions in the catch biomass of all fished stocks (excluding 'other' fisheries). Furthermore, a decrease in rock lobster fishing to combat urchin biomass was predicted to be insufficient to achieve sustainability in all fisheries as all catch biomasses were predicted to decrease following the simulated reduction in rock lobster fishing. Investigation into the implementation of multiple simultaneous management strategies including collaboration between a number of fisheries, each affected by an increase in urchins, may be warranted to determine a method to deal with a substantial increase in urchins. Such approaches are, as yet, few and far between in Australia and worldwide (Pitcher et al. 2009) yet, may be possible through the implementation of EBFM, where the ecosystem can be managed as a whole and other fisheries and species are taken into account during the decision-making process.

The models produced in this study were particularly valuable in highlighting potential responses to change within the ecosystem and to examine a process through which qualitative and quantitative models may be used in conjunction. Yet, the validation of ecosystem model results would have been useful to gauge the applicability of model performance in this study (Rykiel 1996). All available fisheries and survey data, excluding that from marine reserves, were utilised to produce the Ecopath with

Ecosim model. As a result there were no additional data with which to validate the model. The comparison of results with survey data from marine reserves would not adequately validate the model as the small spatial scale (<15km^2, Maria Island Marine Reserve) and low number of reserves may have confounded these analyses. While validation of model results was not undertaken, the Ecopath with Ecosim model remains useful if the results are considered semi-quantitatively. The results may then be used to determine the direction of ecosystem response to perturbation (Okey et al. unpublished); arguably the most important finding of perturbation analysis.

The use of qualitative models prior to Ecopath with Ecosim provided a strategic approach for ecosystem investigation through the generation of predictions and identification of topics of study. This technique was beneficial as it allowed the restriction of the potentially overwhelming number of paths of investigation prior to more detailed quantitative analyses. This capacity to focus ecosystem analyses when many paths of investigation are possible determines this technique may also be useful in data-rich systems. In addition, the capacity to generate predictions using limited data determines that qualitative models can be beneficial in the management of data-poor systems to provide a starting point for further analyses. This case study has provided an example of the process through which complementary models may be used to assess model results and may be useful in both data-poor and data-rich ecosystems. In addition, this study highlighted a number of important issues for management and benefits surrounding the use of qualitative modelling and Ecopath with Ecosim in ecosystem investigations.

Appendix 6.1. Weighted predictions (W) matrix for the qualitative trophic model (Chapter 5). The weighted predictions for the impact of wrasse on banded morwong/bastard trumpeter (BMW/BTR, dashed line) and banded morwong/bastard trumpeter on wrasse (full line) are highlighted.

W=

	1.	*2.*	*3.*	*4.*	*5.*	*6.*	*7.*	*8.*	*9.*	*10.*	*11.*
1.*Invertebrates*	0.87	0.32	0.14	0.10	1	0.57	0.26	0.10	0.87	1	0.077
2.*Wrasse*	0	0.40	0.21	0.20	1	0.32	0.29	0.20	0	1	0.14
3.*Bivalves*	0.43	0.21	0.44	0.17	1	0.12	0	0.17	0.43	1	0.26
4.*Marblefish*	0.31	0.20	0.17	0.23	1	0.081	0.29	0.28	0.31	1	0.091
5.*Ophiuroids*	1	1	1	1	0.25	1	1	1	0.25	0.25	1
6.*Decapods*	0.24	0	0.36	0.24	1	0.57	0.22	0.24	0.24	1	0.20
7.*Seals*	0.26	0.097	0.24	0.097	1	0.073	0.31	0.26	0.26	1	0.31
8.*Algae*	0.31	0.20	0.17	0.28	1	0.081	0.29	0.31	0.31	1	0.091
9.*Detritus*	1	1	1	1	1	1	1	0.25	0.25	1	1
10.*LSB*	1	1	1	1	0.25	1	1	0.25	0.25	0.25	1
11.*BTR/BMW*	0.23	0.43	0	0.27	1	0.20	0.13	0.23	0.23	1	0.42

Appendix 6.2. Diet matrix for the inshore reef ecosystem showing proportions of dietary items for each ecosystem member/group.

Prey/ predators	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18
1.Pelagic sharks																		
2.Marine mammals	0.110																	
3.Seabirds	0.010	0.009																
4.Small sharks + rays	0.300																	
5.Large omnivorous fish	0.025	0.020	0.050				0.030											
6 Small omnivorous fish			0.020	0.028			0.055	0.040										0.007
7.Large piscivorous fish	0.135	0.129	0.065															
8.Small piscivorous fish			0.010	0.009	0.005		0.055	0.150										0.012
9.Large planktivorous fish	0.110	0.048	0.030	0.012			0.040											0.003
10.Small planktivorous fish			0 035	0.034	0.010		0.068	0 150										0.027
11.Large herbivorous fish	0.008	0.054	0.017				0.120											0.019
12.Small herbivorous fish		0.005	0.015	0.052	0.005		0.017	0.040										0.005
13.Juvenile banded morwong			0.001	0.009			0.001											0 002
14.Adult banded morwong	0.015	0.002	0.001	0.002														
15.Bastard trumpeter	0.020	0.002	0 001	0.010														
16.Wrasse	0.035	0.002	0 002	0.015				0.003										0.010
17.Long-snouted boarfish		0 001	0.001	0 001														
18.Cephalopods		0.400	0.146	0 110	0.002		0.017											0.031
19.Rock Lobster				0.043														0.001
20.Decapods		0.030	0.061	0.050	0.100	0 080		0.025					0.060	0.270	0.300	0.081		0.100
21.Other filter feeders					0.027	0.050	0.028						0.002	0.001	0.009	0.058	0.890	
22.Bivalves			0.003	0.004	0.007	0.010		0.010					0.173	0.090	0.022	0.023		0.002
23.Centrostephanus rodgersii				0.006	0.002	0.009												
24.Echinoids			0.028	0.030	0.033	0.041	0.035	0.030					0.017	0.062	0 019	0.052		0.002
25.Asteroids					0.031										0.007			
26.Carnivorous gastropods			0.037	0.016	0.058	0.080		0.024					0 019	0.014	0.135	0.080	0.004	
27.Polychaetes + detritvorous invertebrates				0.047	0 058	0.100		0.016	0.009				0.003	0.051	0.010	0.002	0 006	
28.Herbivorous gastropods			0.006	0.017	0.032	0.130		0.023	0 050				0.012	0.012	0.009	0.038	0 005	
29.Abalone					0.021									0.009		0.001		
30 Small crustaceans			0.004	0 329	0.300	0.400	0.063	0.030	0.243	0.050			0.714	0.445	0.218	0 313	0.005	0.222
31.Zooplankton					0.007			0.010	0.450	0.600					0.100			
32.Green macroalgae											0.295	0 215						
33.Red macroalgae											0.350	0.450						
34 Brown macroalgae					0.159						0.355	0 235						
35.Crustose algae												0.100						
36.Phytoplankton									0.020	0.100								
37.Detritus				0.124		0.100								0.003	0.171	0.022	0.090	0.158
38.Import	0.232	0.298	0.467	0.052	0.143		0.471	0.449	0.228	0.250				0.043		0.330		0.399

Appendix 6.2 (cont.)

Prey/ predators	19	20	21	22	23	24	25	26	27	28	29	30	31	References (see App. 6.3)
1.Pelagic sharks														1, 2
2 Marine mammals														3, 4
3.Seabirds														5, 6, 7
4.Small sharks + rays														8, 9
5.Large omnivorous fish														9, 10
6.Small omnivorous fish		0.002												9
7.Large piscivorous fish														11, 9
8.Small piscivorous fish														11, 9
9.Large planktivorous fish														11, 9
10.Small planktivorous fish		0.005												12, 9
11.Large herbivorous fish														13, 14
12.Small herbivorous fish		0.003												13
13.Juvenile banded morwong														13
14.Adult banded morwong														13
15.Bastard trumpeter														13
16.Wrasse														13
17.Long-snouted boarfish														13
18.Cephalopods														15
19.Rock Lobster														16
20.Decapods	0.019	0.001												17, 18
21 Other filter feeders		0.006					0.113	0.070						19
22.Bivalves		0.010					0.025	0.109						19
23.Centrostephanus rodgersii		0.016					0.007							20
24.Echinoids	0.025	0.039					0.054	0.019						20
25.Asteroids		0.019												21
26.Carnivorous gastropods	0.055	0.027					0.080	0.005						22
27 Polychaetes + detritvorous invertebrates		0.011					0.015	0.050				0.030		19
28 Herbivorous gastropods	0.042	0.001					0.065	0 150						22
29.Abalone	0.030	0 014						0.034						22
30.Small crustaceans	0 210	0.032					0.324	0.079	0.035					24
31.Zooplankton		0.055	0.400	0.100								0.040	0.150	24
32.Green macroalgae		0.040						0.004		0 100	0.005			
33.Red macroalgae		0.040			0.220	0.220		0.013		0.345	0.740			
34 Brown macroalgae		0.050			0.480	0.480		0.013		0.400	0.100			
35.Crustose algae		0.020			0.150	0.150				0.150	0.150			
36.Phytoplankton		0.010	0.400	0.360					0.400			0.190	0.750	
37.Detritus	0.339	0.462	0.200	0.540	0.150	0.150	0.267	0.260	0.565	0.00500	0.005	0.640		
38.Import	0.280	0.137					0.050	0.194				0.100	0.100	

Appendix 6.3 Reference list from Appendix 1.

Reference number from App. 6.2	Reference
1	Shark Advisory Group (1998) Australian shark assessment report for the Australian national plan of action for the conservation and management of sharks. In. Department of Agriculture, Fisheries and Forestry
2	Bruce, B. D. (1992). Preliminary observations on the biology of the white shark, *Carcharodon carcharias*, in South Australian waters. *Australian Journal of Marine and Freshwater Research*, **43,** 1-11.
3	Hume F, Hindell MA, Pemberton D, Gales R (2004) Spatial and temporal variation in the diet of a high trophic level predator, the Australian fur seal (*Arctocephalus pusillis doriferus*). *Marine Biology* **144,** 407-415.
4	Pauly, D., Trites, A. W., Capuli, E. and Christensen, V. (1998). Diet composition and trophic levels of marine mammals. *ICES Journal of Marine Science*, 55, 467-481.
5	Bunce A, Norman FI (2000) Changes in the diet of the Australasian gannet (*Morus serrator*) in response to the 1998 mortality of pilchards (*Sardinops sagax*). *Marine and Freshwater Research* **51,** 349-353.
6	Hedd A, Gales R (2001) The diet of shy albatrosses (*Thalassarche cauta*) at Albatross Island, Tasmania. Journal of Zoology 253:69-90
7	Chiaradia A, Costalunga A, Knowles K (2003) The diet of little penguins (*Eudyptula minor*) at Phillip Island, Victoria, in the absence of a major prey- pilchard (*Sardinops sagax*). *Emu* **103,** 43-48.
8	Treloar MA, Laurenson LJB (2005) Preliminary observations on the reproduction, growth and diet of *Urolophus cruciatus* (Lacepede) and *Urolophus expansus* (McCulloch), Urolophidae, in southeastern Australia. *Proceedings of the Royal Society of Victoria* **117,** 341-347.
9	Bulman CM, Althaus F, He X, Bax NJ, Williams A (2001) Diets and trophic guilds of demersal fishes of the south-eastern Australian shelf. *Marine and Freshwater Research* **52,** 537-548.
10	Edgar G, Shaw C (1995) The production and trophic ecology of shallow-water fish assemblages in southern Australia. II. Diets of fishes and trophic relationships between fishes and benthos at Western Port, Victoria. *Journal of Experimental Marine Biology and Ecology* **194,** 83-106.
11	FishBase, www.fishbase.org
12	MacLeod, D. (2005). Ecological and functional equivalence in small pelagic fishes off the east coast of Tasmania, *School of Zoology,* Hons. thesis, University of Tasmania, Hobart, 72 pp.
13	Metcalf, S. J., Dambacher, J. M., Hobday, A. J. and Lyle, J. M. (2008). Importance of trophic information, simplification and aggregation error in ecosystem models. *Marine Ecology Progress Series*, **360,** 25-36.
14	Clements KD, Choat JH (1997) Comparison of herbivory in the closely-related marine fish genera *Girella* and *Kyphosus*. *Marine Biology* **127,** 579-586.
15	Grubert MA (1996) The reproductive biology, ecology and diet of the maori octopus (*Octopus maorum*) in Eaglehawk Bay, south-east Tasmania. Department of Zoology. University of Tasmania, Hobart, 105 pp.
16	K. Redd and H. Pederson (unpublished), University of Tasmania
17	Gasalla MA, Rossi-Wongtschowski CLDB (2004) Contribution of ecosystem analysis to investigate the effects of changes in fishing strategies in the South Brazil Bight coastal ecosystem. *Ecological Modelling* **172,** 283-306.
18	Manickchand-Heileman, S., Mendoza-Hill, J., Lum Kong, A. and Arocha, F. (2004). A trophic model for exploring possible ecosystem impacts of fishing in the Gulf of Paria, between Venezuela and Trinidad. *Ecological Modelling*, **172,** 307-322.
19	Okey, T. A., Banks, S., Born, A. F., Bustamante, R. H., Calvopina, M., Edgar, G. J., Espinoza, E., Farina, J. M., Garske, L. E., Reck, G. K., Salazar, S., Shepherd, S., Toral-Granda, V. and Wallem, P. (2004). A trophic model of a Galapagos subtidal rocky reef for evaluating fisheries and conservation strategies. *Ecological Modelling*, **172,** 383-401.
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21	Ross DJ, Johnson CR, Hewitt CL (2002) Impact of introduced seastars *Asterias amurensis* on survivorship of juvenile commercial bivalves *Fulvia tenuicostata*. *Marine Ecology Progress Series* **241,** 99-112.
22	Ortiz M, Wolff M (2002) Spatially explicit trophic modelling of a harvested benthic ecosystem in Tongoy Bay (central northern Chile). *Aquatic Conservation: Marine and Freshwater Ecosystems* **12,** 601-618.
23	Shepherd SA (1975) Distribution, habitat and feeding habits of abalone. *Australian Fisheries* **34,** 1-4.
24	Bulman CM (2005) NWSJEMS Milestone Report, Task 2.6. Trophic webs and modelling of the North West Shelf. In. CSIRO, Marine Research, Hobart, p 92pp

Appendix 6.4 Balanced Ecopath group parameters for the inshore reef ecosystem. Values in bold were calculated by Ecopath. Abbreviations: P/B -production/biomass; Q/B -consumption/biomass; EE-ecotrophic efficiency; and P/Q- production/consumption. Large fish >50cm and small fish <50 cm.

Group name	Trophic level	Biomass (t/km^2/ year)	P/B (/year)	Q/B (/year)	EE	P/Q (/year)	References
Pelagic sharks	**4.35**	0.0150	0.200	1.433	**0.000**	0.140	Shark Advisory Group 1998, Bulman et al. 2006
Marine mammals	**3.97**	0 110	0.070	15.000	**0.307**	0 005	Goldsworthy et al. 2003, Bulman et al. 2006
Seabirds	**3.83**	0.188	1.000	25.000	**0.080**	0.040	Fulton and Smith 2002
Small sharks + rays	**3.21**	1.146	0.270	5 500	**0.182**	0.049	FishBase, Bulman et al. 2006
Large omnivorous fish	**2.94**	5.588	0.320	7.920	**0.999**	0.040	Okey et al. 2004b
Small omnivorous fish	**3.04**	4.163	0.470	4.130	**0.812**	0.114	Okey et al. 2004b, Bulman et al. 2006
Large piscivorous fish	**3.56**	4.294	0.418	2.500	**0.628**	0.167	Fulton and Smith 2002
Small piscivorous fish	**3.36**	1.376	0.821	2.737	**0.993**	0.300	Fulton and Smith 2002
Large planktivorous fish	**3.09**	3.000	0.300	2 400	**0.965**	0.125	Bulman et al. 2006
Small planktivorous fish	**3.00**	2.183	1.500	3.300	**0.736**	0.455	Okey et al. 2004b
Large herbivorous fish	**2.00**	2.234	0.780	16 399	**0.962**	0.048	Okey et al. 2004a
Small herbivorous fish	**2.00**	1.586	0.880	25.830	**0.997**	0.034	Okey et al. 2004a
Juvenile banded morwong	**3.12**	0.348	0 310	7.279	**0.844**	0.043	Ziegler et al. 2007
Adult banded morwong	**3.14**	4.587	0.240	2.900	**0.984**	0 083	Bulman et al. 2006, Ziegler et al. 2007
Bastard trumpeter	**3.08**	3 120	0.320	3.000	**0.072**	0.107	Metcalf et al. 2008, Palomares and Pauly 1998
Wrasse	**3.16**	7.224	0.360	3.000	**0.990**	0.120	Metcalf et al. 2008, Palomares and Pauly 1998
Long-snouted boarfish	**3.28**	0.229	0.360	7.600	**0.930**	0.047	FishBase, Okey et al. 2004b
Cephalopods	**2.93**	4.114	1.369	8.000	**0.626**	0.171	Fulton and Smith 2002
Rock Lobster	**2.62**	14.708	0.550	7.400	**0.880**	0 074	Okey et al. 2004a
Decapods	**2.35**	7.608	2.300	10.000	**0.988**	0.230	Blanchard et al. 2002, Bulman et al. 2006
Other filter feeders	**2.46**	5.351	2.800	11.800	**0.999**	0.237	Fulton and Smith 2002
Bivalves	**2.12**	5.116	3 150	23.000	**0.985**	0.137	Fulton and Smith 2002,
Centrostephanus rodgersii	**2.00**	3.804	1.300	3.700	**0.646**	0 351	Okey et al. 2004b
Echinoids	**2.00**	11.266	1.500	3.700	**0.958**	0.405	Okey et al. 2004b
Asteroids	**2.88**	10.245	1.600	4.000	**0.272**	0.400	Trites et al. 1999
Carnivorous gastropods	**2.71**	7.235	2.700	12.000	**0.967**	0.225	Okey et al. 2004a
Polychaetes + detrit. invertebrates	**2.04**	6.272	3.800	22.000	0.961	0.173	Bundy 2001
Herbivorous gastropods	**2.00**	8.938	2.000	11.680	**0.967**	0.171	Fulton and Smith 2002
Abalone	**2.00**	22.557	1.100	12.410	**0.870**	0 089	Fulton and Smith 2002
Small crustaceans	**2.09**	**16.189**	8.005	25.544	0.800	0.313	Okey et al. 2004b
Zooplankton	**2.16**	35.000	18.315	**84.657**	0.800	**0.216**	K. Swadlng pers. comm.
Green macroalgae	**1.00**	**5.854**	20.000	-	0.300	-	Fulton and Smith 2002
Red macroalgae	**1.00**	**48.531**	20.000	-	0 300	-	Fulton and Smith 2002
Brown macroalgae	**1.00**	**21.870**	20.000	-	0 300	-	Fulton and Smith 2002
Crustose algae	**1.00**	**7.556**	23.700	-	0.400	-	Okey et al 2004a
Phytoplankton	**1.00**	**10.991**	368.000	-	0.600	-	SeaWiFS, Bulman et al. 2006
Detritus	**1.00**	100.000	-	-	0.140	-	Bulman et al. 2006

7. GENERAL DISCUSSION

In Australia, the Commonwealth Government and State Government of Western Australia, as well as other countries such as the USA, Canada and Norway, have made some progress towards the implementation of EBFM (Pitcher et al. 2009). Despite this progress, calls for the process of implementation to be accelerated in data-limited situations have been made (Fletcher 2003, Forrest 2003). As a result, the primary aim of this thesis was to investigate methods of analysis that could be used to assist the implementation of EBFM in data-limited situations. As EBFM becomes more widely accepted and used it will become critical for the management of Tasmanian fisheries to follow suit or potentially face a loss of accreditation. In addition, the need to implement EBFM will become more important if ecosystem-level changes, that are unaccounted for in current management strategies, increase the risk of fishery collapse. This thesis also aimed to highlight ecosystem dynamics and perturbations that impact Tasmanian banded morwong populations, the fishery for banded morwong and the associated inshore reef ecosystem. The findings of this thesis should be used to guide further studies in the region to aid sustainable fisheries management in Tasmania.

7.1 Summary of analyses for EBFM

A systematic approach to ecosystem analyses for the banded morwong fishery and associated ecosystem were described. In data- and resource-limited situations this may require some reanalysis of existing data sources (Chapter 2) or primary data collection (Chapters 4 and 5) to assess multi-species interactions and ecosystem dynamics. These interactions and dynamics are important as the exploitation of fish stocks can indirectly impact non-target species and other fisheries as well as significantly alter ecosystem structure and functioning (Thrush et al. 1995, Larkin 1996, Roughgarden and Smith 1996, Auster and Langton 1999, Pauly et al. 2001, Christensen et al. 2003). Ignoring potentially important fishery effects, such as bycatch mortalities, and underutilising the available data may determine that ecosystem analyses are not sufficient to identify critical responses to change and are therefore of little use to management. This thesis was structured so as to allow the identification of important relationships and the prioritisation of further investigation using data analysis and primary data collection. This process allowed the construction of models for the analysis of ecosystem dynamics and the discussion of potential management strategies for the inshore reef ecosystem of eastern Tasmania. Importantly, the issues identified, such as model assumptions and uncertainty, can provide guidance for other ecosystem studies in data-limited situations as to how to improve analyses in the future.

The definition of the spatial scale of the study was undertaken as the initial step in the analysis of the inshore reef ecosystem and was achieved through the examination of available fishery and biodiversity data. Key ecosystem constituents were identified, including commercially important fish as well as dominant invertebrate and algal species. The identification of these key species aided the development of qualitative and quantitative ecosystem models (Chapters 3, 5 and 6), focussed data collection (Chapters 4 and 5) and enabled the aggregation of species into groups for the Ecopath with Ecosim model (Chapter 6). The analysis of available data to determine the appropriate spatial scale and key species is critical to bound ecosystem studies (Thrush 1999, Olivier and Wotherspoon 2005) when data is limiting, providing a limit to the perturbations, species and dynamics that require investigation. Data-limited situations require these bounds to ensure the task of ecosystem analysis is achievable and able to be directed using the small amount of information available. For instance, if the only available data shows that a small-scale trap fishery is impacting an inshore reef and all other potential perturbations are unknown, restricting the spatial scale of the research may make additional data collection more viable. Producing spatial bounds is also a method of reducing the uncertainties involved in analyses (Olivier and Wotherspoon 2005) by restricting the number of perturbations and species involved in ecosystem dynamics.

The second step in the process of ecosystem analysis was the production of qualitative models to investigate general ecosystem dynamics and relationships (Chapter 3). The qualitative models were used to produce predictions of response to perturbations, which were integral in guiding and focussing the subsequent research questions. Although the identification of research topics could have been undertaken using pre-existing knowledge of the ecosystem and fisheries, qualitative models provided a strategic way to investigate research priorities and focus further investigations by reducing the number of paths of investigation. In addition, the production of predictions of response provided alternative results with which to investigate the robustness of conclusions. The formulation of hypotheses may be difficult in data-limited situations due to the seemingly overwhelming complexity of system dynamics. Yet, methods, such as qualitative modelling, can be used to simplify and focus thought processes as well as formulate hypotheses for further research. Other ecosystem models, such as the trophodynamic model Ecopath with Ecosim, have been used for a similar purpose when a larger amount of data is available (Tsehaye and Nagelkerke 2008). The benefit to using qualitative modelling in data-limited situations is that this technique can utilise trophic and non-trophic information as well as any trends, opinions and quantitative data sets that may be available.

The identification of important species and relationships using data analysis and qualitative models can highlight data gaps that require investigation prior to further analyses. This is a critical step, particularly in data-limited situations, as it can focus data collection to these gaps in knowledge (Caruso and Ward 1998). For instance, fishery effects can extend beyond the species harvested and in the case of non-target species, can involve incidental fishery-induced mortality (Broadhurst et al 2006). Knowledge of the non-target species captured as well as their mortality rates is necessary to gauge the overall impact of the fishery. This is particularly the case with gillnet fisheries, such as the banded morwong fishery. Information on the survival rates of discarded individuals from the key species captured by the banded morwong fishery was unavailable. This information, in addition to information on catch and survival rates for other fisheries (when available), was considered necessary to allow the production of the Ecopath with Ecosim model and was examined using the relationship between condition at capture and short-term survival rates (Chapter 4).

Detailed ecosystem-specific trophic information was also identified as a data gap for the majority of the commercially important fish species. Ecopath with Ecosim is a trophodynamic model (Walters et al. 1997) and required dietary information for all groups in the model. Dietary information was collected for six commercially important fish species and used to create a qualitative trophic model for the initial assessment of trophic dynamics. As the decision of whether or how far to simplify models is often an important step in ecosystem analyses (Raick et al. 2006), different aggregation techniques were investigated using this qualitative model. A decision on the appropriate technique to use was made based on aggregation error (Auger et al. 2000), which is critical in order to ensure predictions are useful. The acknowledgement of and response to uncertainty (or error) has also been identified as an important element in the implementation of EBFM (Marasco et al. 2007). The infrequently used aggregation technique, regular equivalence (Luczkovich et al. 2003), was found to create the least aggregation error in the qualitative models (Chapter 5) and was suggested for use in the simplification of future trophic studies. Such findings will be of benefit in data-limited ecosystem studies as the collection of trophic information on all species would be impossible.

Quantitative models, such as Ecopath with Ecosim and Multispecies Virtual Population Analysis (MSVPA), have been used in the large majority of recent ecosystem analyses. These models are beneficial to guide ecosystem analyses and management as they can provide more precise estimates of change due to perturbation than is possible using conceptual or qualitative models (Levins 1966). This precision is necessary to allow the production of detailed management strategies (Nicholson and Jennings 2004). For instance, the Ecopath with Ecosim model identified the need for a reduced TAC in the banded morwong fishery as well as the monitoring of primary production, urchin and decapod abundance. In addition, the need to continue the collection of fishing rate (to estimate effort) and catch biomass (weight of captured species) data was identified. Furthermore, the impact of a substantial increase in urchins was found to be considerable, even following a reduction in rock lobster fishing rate. The simulations used to investigate the ecosystem response to an increase in urchins and rock lobster (following a reduction in fishing) showed that increased rock lobster abundance would be insufficient to allow catch biomasses of fished species and overall biomasses of functional groups to increase. In other words, a simple reduction in rock lobster fishing to allow greater predation on urchins would not be sufficient to ensure the sustainability of multiple fisheries nor compensate for the impact of urchin barrens on the broader ecosystem. The simultaneous modification of multiple fisheries management strategies was suggested for further analysis.

The use of quantitative models in data-limited situations can be enhanced through the use of alternative models. Different models can provide alternative views of ecosystem dynamics and processes by maximising different combinations of realism, precision and generality (Levins 1966). Consistent conclusions between models, with different uncertainties and assumptions, suggest the claim is robust (Levins 1993) for that particular system. For example, the predictions that a TAC for banded morwong can have a stabilising effect on the system (if set at an appropriate level) and that rock lobster can control urchin abundance, allowing foliose algae to increase, were robust to the differences between models. Conversely, the finding of inconsistent results between different models can also be useful in data-limited situations as it can focus researchers to investigate the cause of the difference, which could include different model assumptions or incorrect model inputs.

7.2 Implications and future directions

Ecosystem models and associated analyses are an integral part of EBFM and the process undertaken in this study may serve to inform decision-making with regard to the potential implementation of EBFM in Tasmania. For instance, the analyses suggested that the TAC for banded morwong should be reduced from the current harvest levels to ensure sustainability. The use of a lower TAC may lessen the likelihood of the over-harvest of banded morwong as well as potential unexpected and undesirable ecosystem effects, such as shifts in community structure as a result of altered trophic links. This may aid the avoidance of substantial ecosystem changes, such as large increases in phytoplankton and jellyfish abundance, which were observed in the Black Sea following years of overfishing (Daskalov 2002). The removal of predators (i.e. fish) from the Black Sea was suggested to have created a trophic cascade and resulted in population explosions of lower trophic-level species (i.e. phytoplankton and jellyfish). The ecosystem models and analyses in the current study, including the qualitative models and the simplification of the trophic web using the REGE algorithm may have been useful in this situation to assess the potential impacts of change due to fishing. More detailed models, such as Ecopath with Ecosim, could then have been utilised to further assess the trophic implications of change.

The methods undertaken in this thesis may be useful in future studies to identify data needs and species or groups that could be used as baseline data for the monitoring of ecosystem change. In addition to a reduction in the TAC, the need for monitoring of primary production in Tasmania as well as urchin and decapod abundance was identified using ecosystem modelling. This baseline data could serve as an early indication of ecosystem change that may have flow-on effects on fisheries sustainability. Managers and fishers may then have time to implement strategies necessary to adapt and ensure fishery sustainability. Such ecosystem indicators have been effective in many systems around the world, particularly with regard to pollution (Phillips 1976), yet are still being discussed for the indication of ecosystem status (Hilty and Merenlender 2000, Carignan and Villard 2002). Indicators of ecosystem status (i.e. whether changes in ecosystem structure and function are occurring) are a critical gap in the current knowledge and could be highly beneficial in the implementation of EBFM by providing an early warning of ecosystem change. Simplified ecosystem modelling methods may allow this work on indicators to progress by providing both a method of clarifying knowledge in complex systems and providing alternative results for the investigation of the robustness of conclusions.

The simulation results in this thesis suggested changes in the ecosystem and harvest rates for key exploited species may have far reaching effects within the focal system. As a result, changes within the ecosystem and fisheries need to be considered concurrently by management in Tasmania. The issue of integrating the management of multiple fisheries is a common factor confronting agencies attempting to implement EBFM (e.g. Costanza et al. 1998, Hanna 1998, Sissenwine and Mace 2002); however the integration of fisheries management is a critical step in the process of holistic management. Investigation into ecosystem dynamics, climate change and other perturbations requires that fisheries effects be considered cumulatively across whole ecosystems. Such a requirement may hamper the investigation of data-limited systems, as information on each perturbation may not be available. In these situations, the use of expert opinion may be necessary to allow investigations to progress. The inclusion of stakeholders from a range of backgrounds (e.g. fishers, fishery managers, conservationists) may require a technique of communication able to be understood from different viewpoints. The simplification of ecosystem structure and dynamics through qualitative modelling can be beneficial as it reduces the complexity of analyses, which can result in an increased understanding of ecosystem dynamics and aid the communication of information between different stakeholder groups (e.g. commercial and recreational fishers, general public, resource managers and scientists). For the implementation of EBFM to be effective, the understanding and acceptance of the management methods by stakeholders is critical (Pomeroy and Berkes 1997). In addition, the inclusion of stakeholders in EBFM may be beneficial. For example, fishers worked together with government agencies to implement appropriate fishing controls in Barbados (Mahon et al. 2003). This strategy provided fishers with greater control over their resources and can, in turn, foster sustainable fishing practices. Yet, with the inclusion of non-scientists in decision-making comes the need for appropriate communication tools in order to avoid non-compliance and adversarial relationships between fisheries and management (Nielsen and Mathiesen 2003, Kaplan and McCay 2004). Qualitative modelling can provide such a tool.

The application of qualitative models to underpin EBFM in the future may benefit from semi-quantitative methods that can assess the magnitude or probability of responses in qualitative models (Hosack et al. 2008). This will provide managers with a gauge for the prioritisation of management strategies based on qualitative predictions. Such methods for assessing the magnitude or probability of predictions will likely include the use of Bayesian Belief Networks in which some initial research has been undertaken with regard to qualitative models (Hosack et al. 2008). In addition, genetic algorithms (Janssen et al. 2000) or fuzzy logic (Paterson et al. 2007) may provide alternative mechanisms for a pseudo-quantitative assessment of qualitative models. The ability to assess qualitative model predictions with techniques, such as these, may also make qualitative models more appealing for use in a broader range of ecological management scenarios.

Ecosystem based fisheries management in Tasmania and elsewhere would also benefit from the inclusion of social and economic trends into decision-making. Social and economic impacts, in addition to ecological impacts, must be taken into account to allow the successful implementation of EBFM (Fletcher 2002). Yet in practice, information on the social aspects of most fisheries in Tasmania is lacking. The identification of important social variables, data collection and analyses of social and economic information should be focussed on in greater detail using qualitative models to determine the most critical data gaps.

As the need for an ecosystem approach to fisheries management increases, due to cumulative human and environmental impacts, such methods of aiding the process of ecosystem investigation are crucial to produce sustainable ecosystems and fisheries. For this reason, confidence in the benefits of qualitative modelling to underpin EBFM has recently been shown in Western Australia. In late 2007, the Department of Fisheries began developing qualitative models to investigate the links between social systems, the economy and ecological communities with respect to state fisheries in a trial EBFM system (Metcalf, unpublished data). Qualitative models were seen as a necessary tool to provide information on social links, in particular, as no quantitative data were available regarding the social impacts of fisheries in the state. As part of a risk assessment approach to implementing EBFM, qualitative modelling is being used as a key tool to identify important relationships that require further investigation, as well as highlighting data gaps and potential indicators of change (sensu Hayes et al. 2008), as undertaken in this thesis. This technique is also being used to aid discussion with non-scientists through the depiction of the ecosystem and its dynamics, which it is hoped will aid the development of EBFM through greater participation of stakeholders and the public.

Despite the progress made in this thesis towards access to and efficiency of ecosystem analyses, there is still significant progress to be made before EBFM can be widely implemented. For example, to enable the broad-scale implementation of EBFM, policy regarding this broadening in management focus is necessary for each jurisdiction (Garcia and Cochrane 2005). Inflexible Fisheries Acts and a lack of authority to make decisions at a local level have been suggested as reasons for the slow uptake of EBFM in the US Caribbean (Appeldoorn 2008). Some changes to policy in Australia have already occurred, including a national approach to ecologically sustainable development (ESD or ecological approach to fisheries, EAF), which takes social, economic and environmental effects into account in management (Fletcher 2002). In addition, fisheries legislation explicitly incorporating non-target and ecological impacts has also been passed (Smith et al. 2007). Similar approaches have also been developed in the North Pacific, Northeast Atlantic and North Sea (Garcia and Cochrane 2005). Nonetheless, there appears to be a gap between these high level policies and the practical implementation of EBFM. In Australia, this is in part due to the need for legislation regarding EBFM to be passed at the state government level. Once the initial framework for the implementation of EBFM is completed in Western Australia, that state will be the first in Australia to develop legislation regarding EBFM. The implementation of EBFM at ground-level will then be possible. A similar process should be undertaken in Tasmania to ensure fisheries, target species and their ecosystems remain sustainable.

The research undertaken in this thesis has shown that techniques such as qualitative modelling can be highly useful tools for fisheries managers and researchers, particularly in data-limited situations. Although the use of complex quantitative models remains an important facet of ecosystem analyses and EBFM, the significant benefits involved in the use of qualitative models should not be ignored. The underlying process of ecosystem analysis undertaken in this thesis can be used to address the practicalities involved in ecosystem analyses in data-limited situations. The process is beginning to be used in Western Australia and will have significant effects on the progression of EBFM in Australia.

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